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SCS Engineers, Long Beach, CA

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Health Effects Associated with Wastewater Treatment and Disposal Systems State-of-the-Art Review

Volume I

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VOLUME I

by

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4014 Long Beach Boulevard
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This report has been reviewed by the Health Effects Research Laboratory, U.S. Environmental Protection Agency, and approved for publication. Approval does not signify that the contents necessarily reflect the views and policies of the U.S. Environmental Protection Agency, nor does mention of trade names or commercial products constitute endorsement or recommendation for use.

FOREWORD

The many benefits of our modern, developing, industrial society are accompanied by certain hazards. Careful assessment of the relative risk of existing and new man-made environmental hazards is necessary for the establishment of sound regulatory policy. These regulations serve to enhance the quality of our environment in order to promote the public health and welfare and the productive capacity of our Nation's population.

The Health Effects Research Laboratory, Research Triangle Park, conducts a coordinated environmental health research program in toxicology, epidemiology, and clinical studies using human volunteer subjects. These studies address problems in air pollution, non-ionizing radiation, environmental carcinogenesis and the toxicology of pesticides as well as other chemical pollutants. The Laboratory participates in the development and revision of air quality criteria documents on pollutants for which national ambient air quality standards exist or are proposed, provides the data for registration of new pesticides or proposed suspension of those already in use, conducts research on hazardous and toxic materials, and is primarily responsible for providing the health basis for non-ionizing radiation standards. Direct support to the regulatory function of the Agency is provided in the form of expert testimony and preparation of affidavits as well as expert advice to the Administrator to assure the adequacy of health care and surveillance of persons having suffered imminent and substantial endangerment of their health.

Pursuant to the overall mission of the laboratory to investigate the human health effects of environmental pollution, this contract was let to summarize the state of knowledge regarding the health effects associated with various methods of handling wastewaters and sewage sludges. Subjects of interest include biological as well as chemical pollutants. Pathways of these agents from wastewater streams to man were delineated.

F. G. Hueter, Ph.D.
Director
Health Effects Research Laboratory

ABSTRACT

This report with its annotated bibliography discusses the scientific knowledge available through 1976 regarding the potential health effects associated with the treatment of wastewater and associated sludges. Also research projects in the area are noted as well as research recommendations. The report is organized to follow the contaminants present in wastewater through the processes used by man to deal with it back to human exposure. Efforts are made to discuss possible health effects resulting from the many processes used.

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SECTION I

SUMMARY AND CONCLUSIONS

EXECUTIVE SUMMARY

The goals of this project were to conduct a state-of-the-art review of existing literature through September 1976 for the following topics:

- Variability in concentration of potential health-impairing constituents in wastewater influent to municipal wastewater treatment plants
- Effectiveness of municipal wastewater treatment plants in reducing health-impairing pollutants through separation into sludges, destruction, or changes in form
- Possible pathways that health-impairing pollutants may take to intersect with man once they are discharged to the biosphere from wastewater and sludge management operations
- Potential public health-related effects from wastewater treatment and disposal management alternatives, including estimated pollutant concentrations associated with possible health impairment
- Future research priorities to better understand the potential impact of wastewater treatment and disposal alternatives on public health.

In reviewing the literature, the possible public health-impairing contaminants were divided into groupings, as follows:

- Water quality parameters, e.g., BOD, SS, nitrogen forms, etc.
- Elemental contaminants, e.g., trace metals
- Biocidal contaminants, e.g., pesticides, biocides
- Synthetic/organic contaminants, e.g., chloro-organics, benzene compounds
- Biological contaminants, e.g., bacteria, virus, and parasites.

Water quality parameters commonly used as a measurement of wastewater treatment effectiveness, such as BOD and suspended solids, have negligible potential effects on public health. These parameters, in themselves, are not good indicators of potential hazards in the discharges from treatment plants. A particular effluent could be acceptable on the basis of low levels of common water quality parameters and yet be potentially hazardous to public health because of high concentrations of elemental or synthetic organic contaminants. Certain water quality parameters, such as BOD, suspended solids, and ammonia may indirectly reduce the efficiency of disinfection treatments and act as carriers of other contaminants. The one contaminant in this classification with potential direct public health effects is nitrate that originates in wastewater and, subsequently, reaches drinking water supplies in high concentrations. This potential hazard has been well known for over 40 yr, and no additional research is recommended.

The elemental contaminants, primarily trace metals, have been well represented in the literature; however, relatively little research has focused on their chemical form or low level health effects. Elemental chemical speciation in the wastewater treatment plant - through the biosphere, and within the human body itself - deserves future research attention. In addition, more information is desirable on the chronic health effects of exposure to trace quantities of elemental contaminants over long periods of time.

The literature reviewed revealed that the input of biocidal contaminants (chlorinated pesticides, herbicides, etc.) to the ecosystem from wastewater treatment and disposal systems is generally negligible when compared to inputs from agricultural runoff and vaporization from surface applications. Further research is recommended concerning locally high biocidal contaminant concentrations originating from wastewater treatment and disposal systems.

The synthetic/organic contaminant grouping, other than pesticides/biocides, includes thousands of organically based compounds produced synthetically or naturally. Little information is available concerning the potential health effects to the public of exposure to these substances in trace quantities over long periods; measurement techniques are often lacking. Many organic chemical manufacturers and industrial users of organic chemicals discharge contaminated wastes into municipal sewage management systems. The fate of the organic compounds during treatment and after various discharge alternatives is largely unknown. It is possible that municipal wastewater treatment plant effluents are a major source or organic contaminants considered potentially carcinogenic or otherwise toxic. This contaminant group will require considerable research funding in future years if the potential health effects of the synthetic/organic chemicals are to be understood.

The final contaminant grouping is the biologicals, which include pathogenic bacteria, viruses, parasitic worms, etc. Traditionally, biological contaminants have received the most attention in wastewater treatment and disposal systems, since these contaminants may directly cause public disease outbreaks. Viral contaminants have not been as extensively studied as other biologicals, because measurement techniques have only been developed in recent years. There remains a considerable amount of research to be accomplished on the basic chemistry of viruses in the wastewater treatment plant, survival in aquatic environments, and survival and transport through soil matrixes when wastewater or sludges are applied on land. In addition, more investigation is needed to measure the transport of aerosolized pathogens from wastewater treatment plants and land application sites.

CONCLUSIONS

INTRODUCTION

The conclusions for this project will be centered around the following contaminant groupings:

- Water quality parameters
- Elemental contaminants
- Biocidal contaminants
- Synthetic organic contaminants
- Biological contaminants

Included for each grouping are a generalized summary and recommendations for future research. Note that the summaries and recommendations pertain only to the contaminants in relation to municipal wastewater treatment and disposal systems. The potential health effects of contaminants from other sources have not been considered. For example, the environmental impact of biocidal contaminants from municipal wastewater systems is generally negligible (except in rare local cases) compared to other sources, e.g., surface water runoff from agricultural areas. Thus, few specific recommendations for future research of the biocidal contaminants are made. However, this must not be misconstrued to mean that knowledge concerning these contaminants is complete, or that the contaminants have no impact on public health.

Watewater treatment and disposal systems may impact certain areas in the biosphere, which in turn may adversely affect public health. To better understand these areas, environmental pathway charts were developed for 20 contaminants that are representative of each contaminant group, as follows:

Contaminant	Contaminant Group
Nitrogen species	Water quality parameters
Arsenic	Elemental
Cadmium	Elemental
Copper	Elemental
Lead	Elemental
Manganese	Elemental
Mercury	Elemental
DDT	Biocidal
Dieldrin	Biocidal
Parathion	Biocidal

Contaminant	Contaminant Group
2, 4-D	Biocidal
PCB's	Synthetic/organic
Choloroform	Synthetic/organic
Polio Virus	Biological
ECHO virus	Biological
Salmonella	Biological
Coliforms	Biological
Escherichia coli	Biological
Protozoa	Biological
Parasitic worms	Biological

The purpose of the environmental pathways charts is to show the relative, potential importance of various transmission pathways leading health-impairing contaminants back to man. Mass balances were derived for contaminant movement through the wastewater treatment plant, through the environment, and finally to the public (see Figure 1). Where possible, existing research was used in developing the mass balances. Lacking specific research, simplifying assumptions were made, and the mass balance figures were calculated from theoretical considerations. The actual calculations are not included in this document, because the myriad of numbers developed would cloud the issue, i.e., those pathways which are critical from the public health standpoint and require additional research consideration in the future. To formulate the critical pathways, "worst case" conditions were assumed in the development of the mass balances. The assumption has been made that public water supplies derived from surface water sources will be adequately treated by conventional methods prior to human consumption.

For each contaminant group discussed, applicable environmental pathways charts are presented. The critical pathways are graphically highlighted with bold lines and arrows. These critical pathways should not be misconstrued as always being deleterious to public health, but rather as being indicative of pathways which may cause public health problems under worst case conditions.

WATER QUALITY PARAMETERS

The water quality parameters considered in the report (BOD, COD, suspended solids, nitrates, phosphates, etc.) are established indicators to predict the effect of wastewater effluents upon the quality of aquatic environments and consequently have been extensively studied. These contaminants offer no significant health hazard to man in concentrations normally found in domestic wastewater treatment plant effluents. Although high general contaminant loadings may adversely affect receiving waters (causing oxygen depletion,

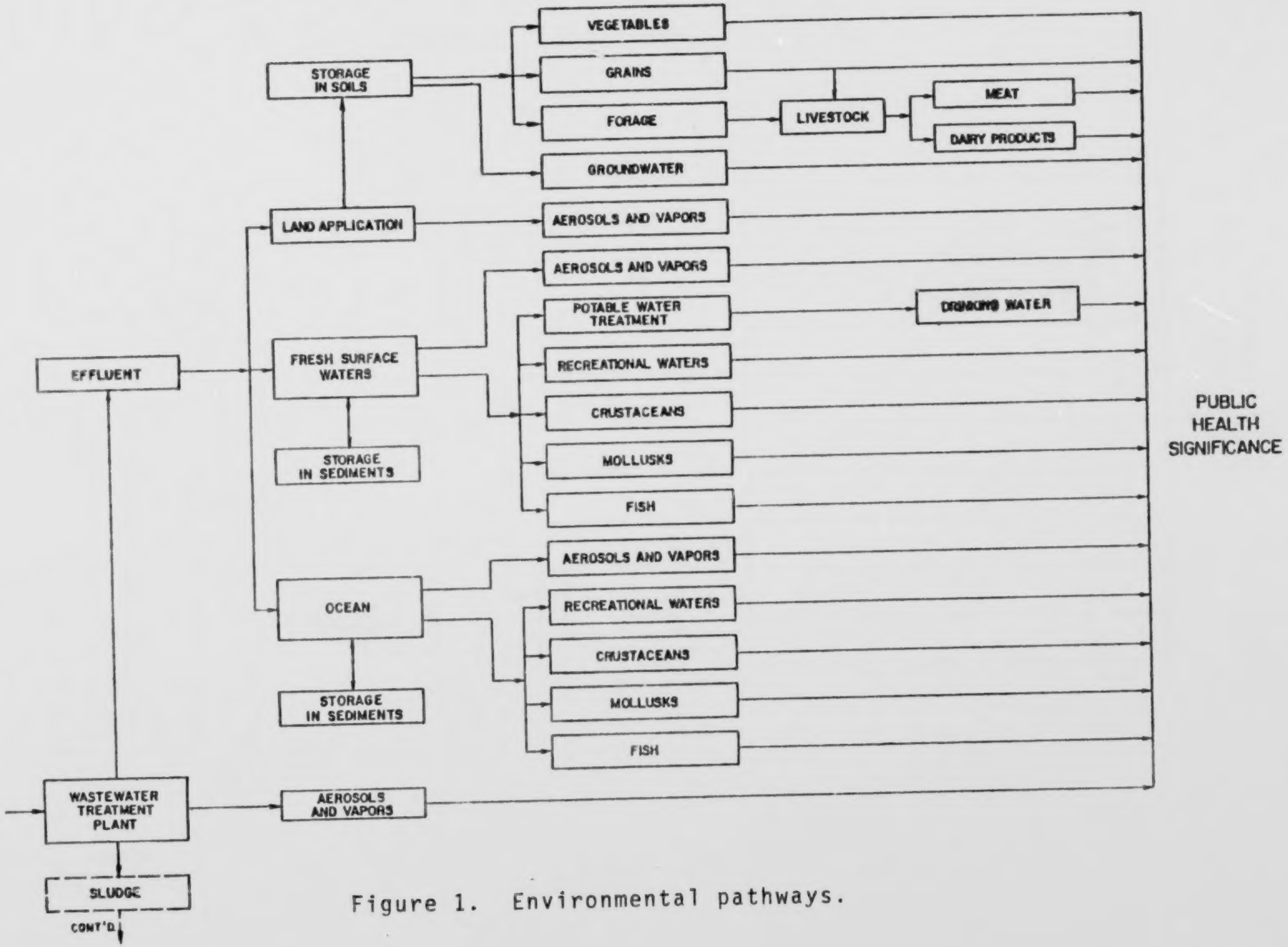


Figure 1. Environmental pathways.

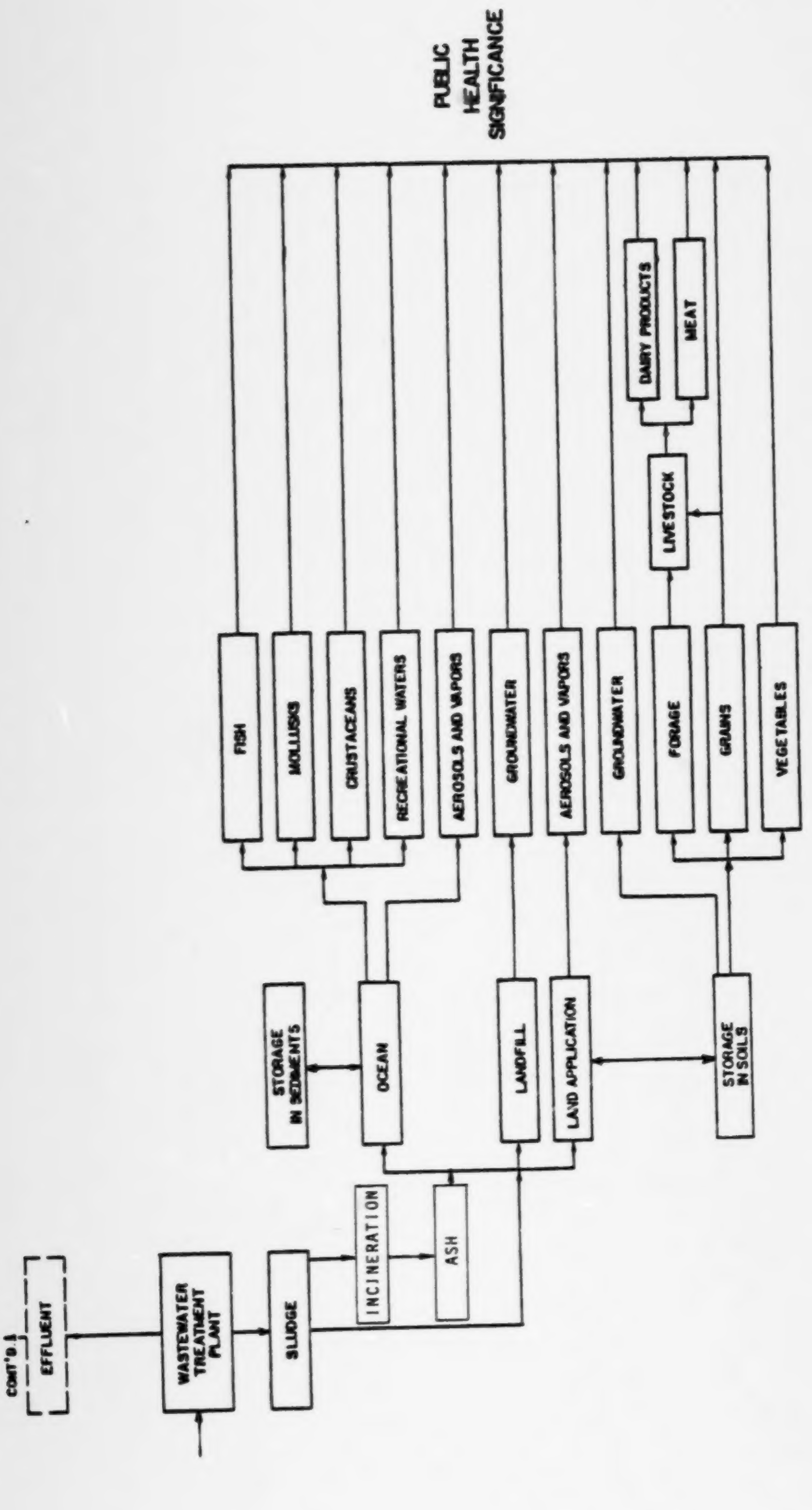


Figure 1. (continued)

eutrophication, etc.), neither they nor the by-products generated under these conditions are considered a health hazard. The only general contaminant prevalent in the literature as a potential health hazard is nitrate in drinking water. The critical pathways of nitrogen species may be seen in Figure 2. The major pathway is through groundwater after disposal of effluent or sludge on land. This topic has been well documented. Much of the present research is concerned with development of regulatory standards for nitrogen application to land in order to determine maximum "safe" levels of annual effluent and/or sludge land application. No further study is recommended beyond the current scope of the current research.

ELEMENTAL CONTAMINANTS

The elemental contaminants examined in this study are:

Aluminum
Antimony
Arsenic
Barium
Beryllium
Boron
Cadmium
Chromium
Cobalt
Copper
Germanium
Iron
Lead
Manganese
Mercury
Molybdenum
Nickel
Selenium
Thorium
Tin
Uranium
Zinc

The acute health effects of most of the elemental contaminants are well documented in the literature. However, little information exists concerning chronic health effects from trace exposures over long periods of time. Most of the elements listed are essential in small amounts for adequate diet. Too little intake may be just as harmful as too much. Thus the problem often lies in defining a healthful range of human intake. The environmental pathways of the elemental contaminants are shown in the following figures:

Contaminant	Figure
Arsenic	3
Cadmium	4
Copper	5
Lead	6
Manganese	7
Mercury	8

An examination of the environmental pathway charts reveals several generalized conclusions. Municipal discharges to fresh

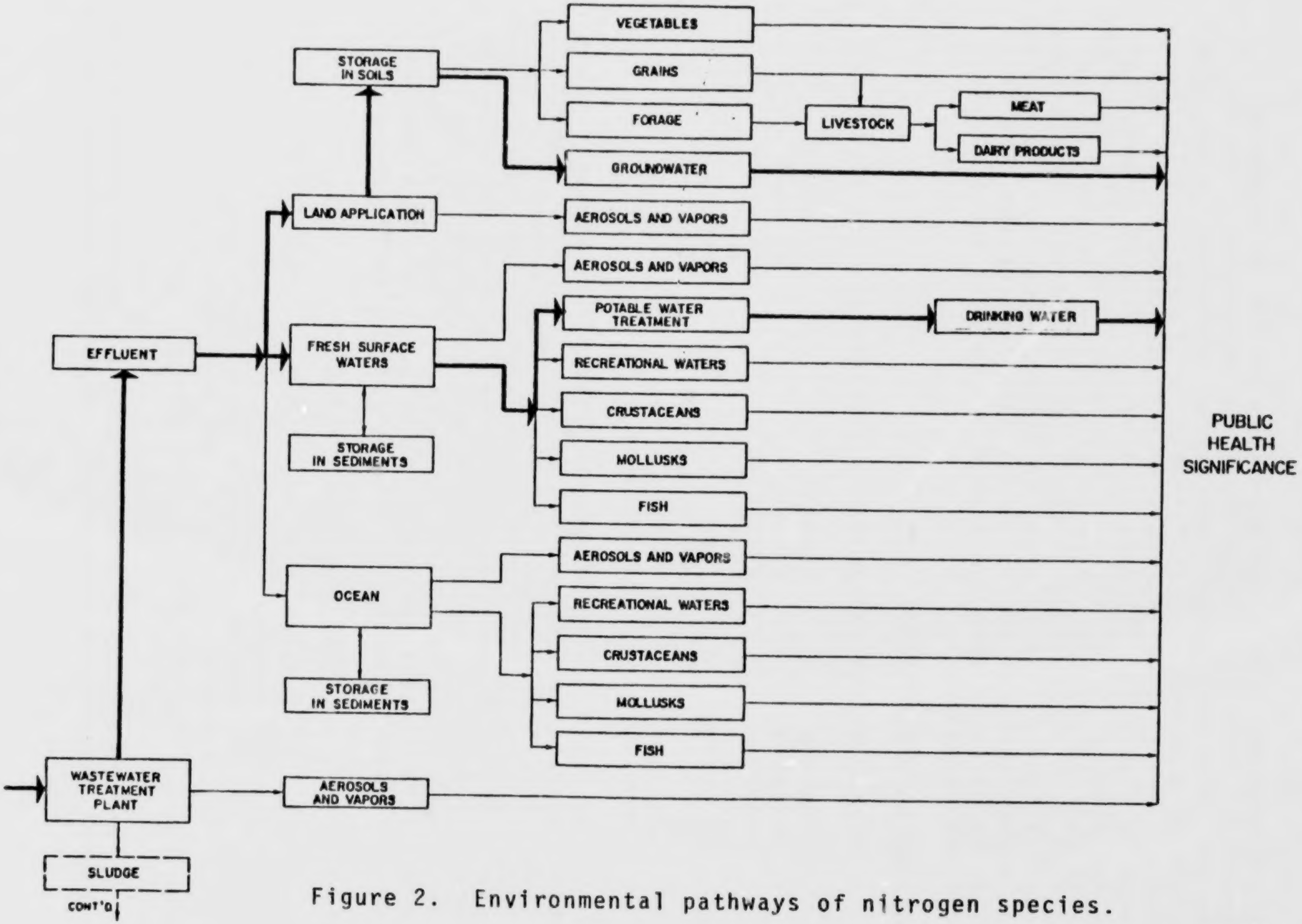


Figure 2. Environmental pathways of nitrogen species.

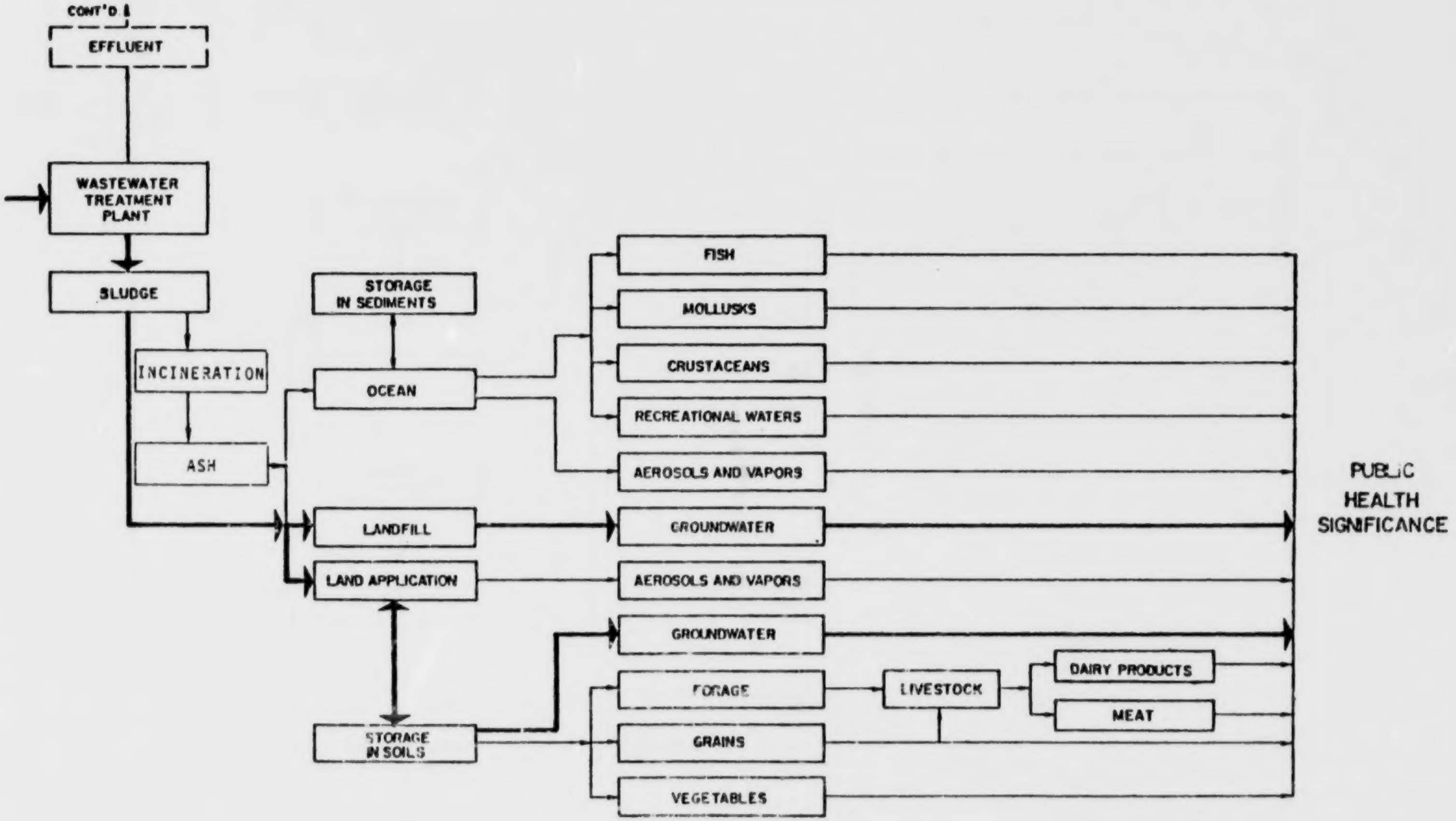


Figure 2. (continued)

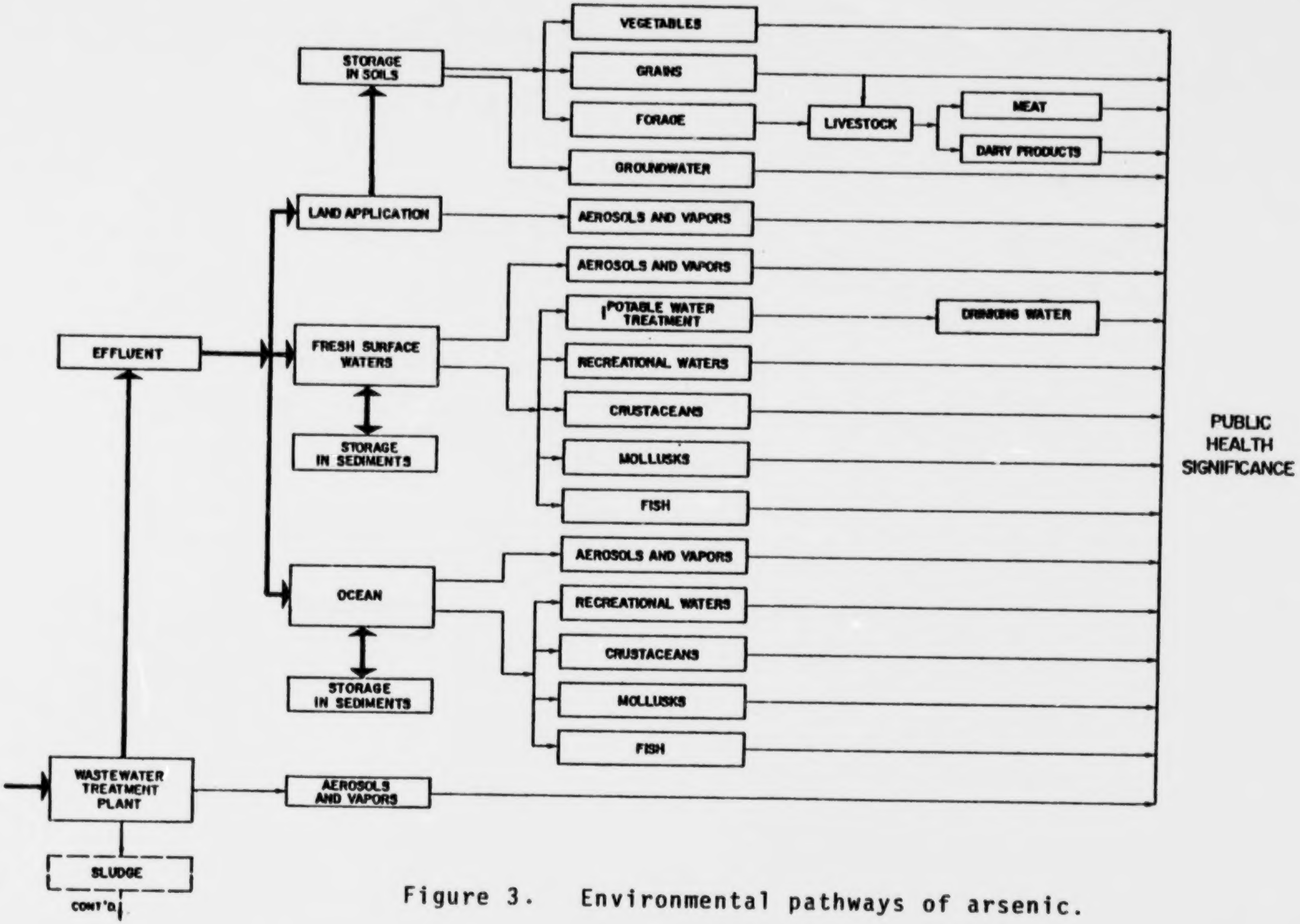


Figure 3. Environmental pathways of arsenic.

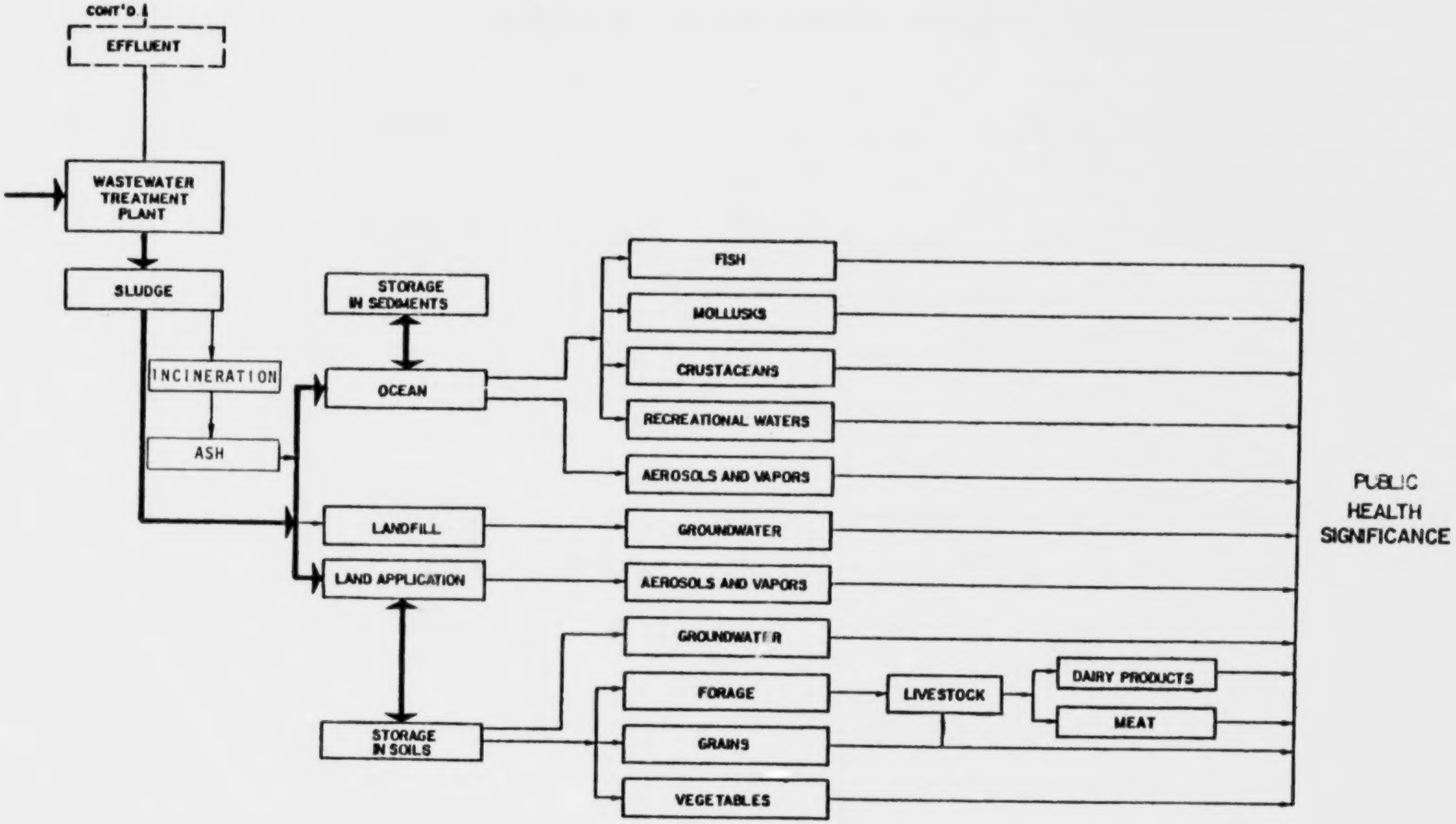


Figure 3. (continued)

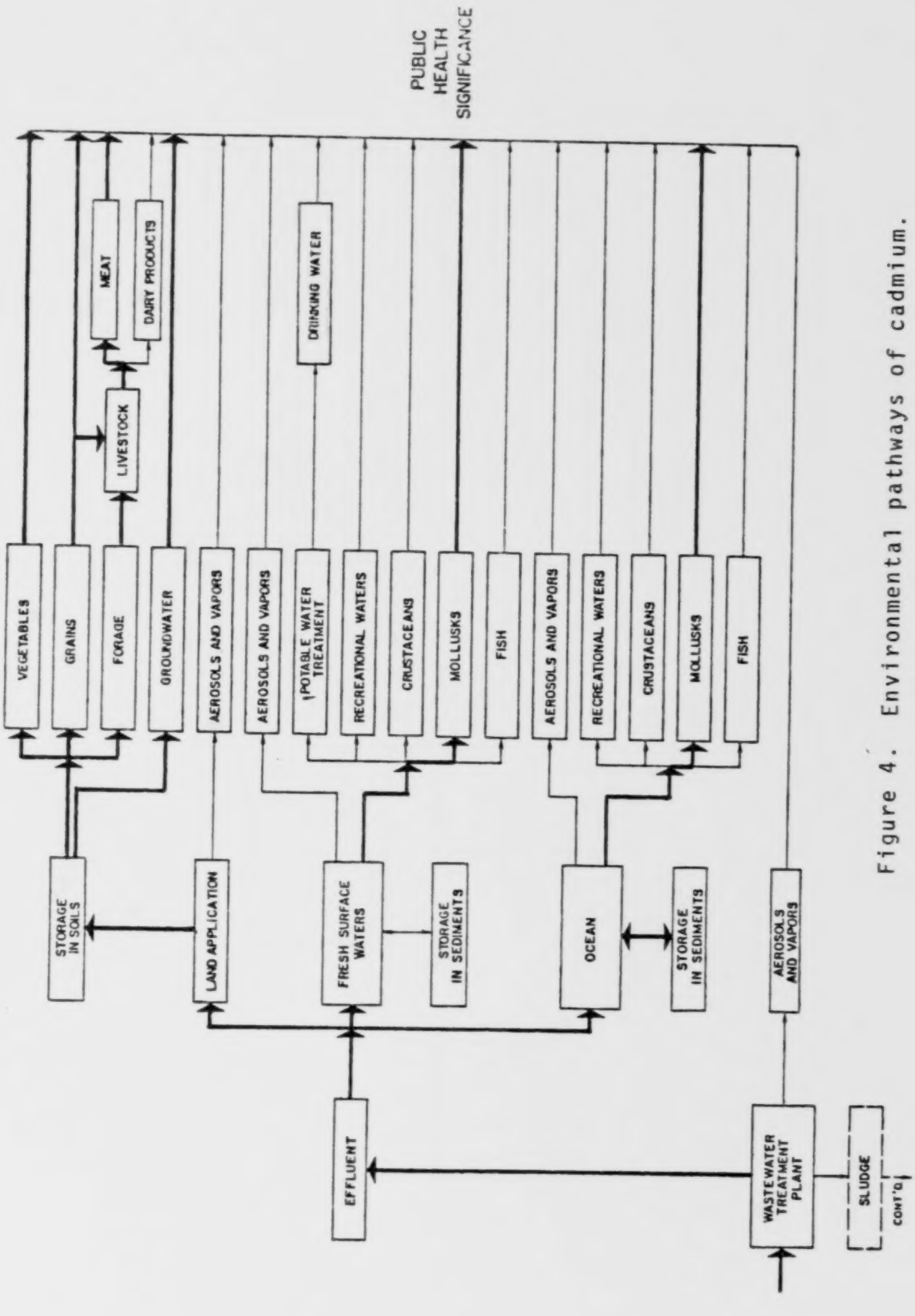
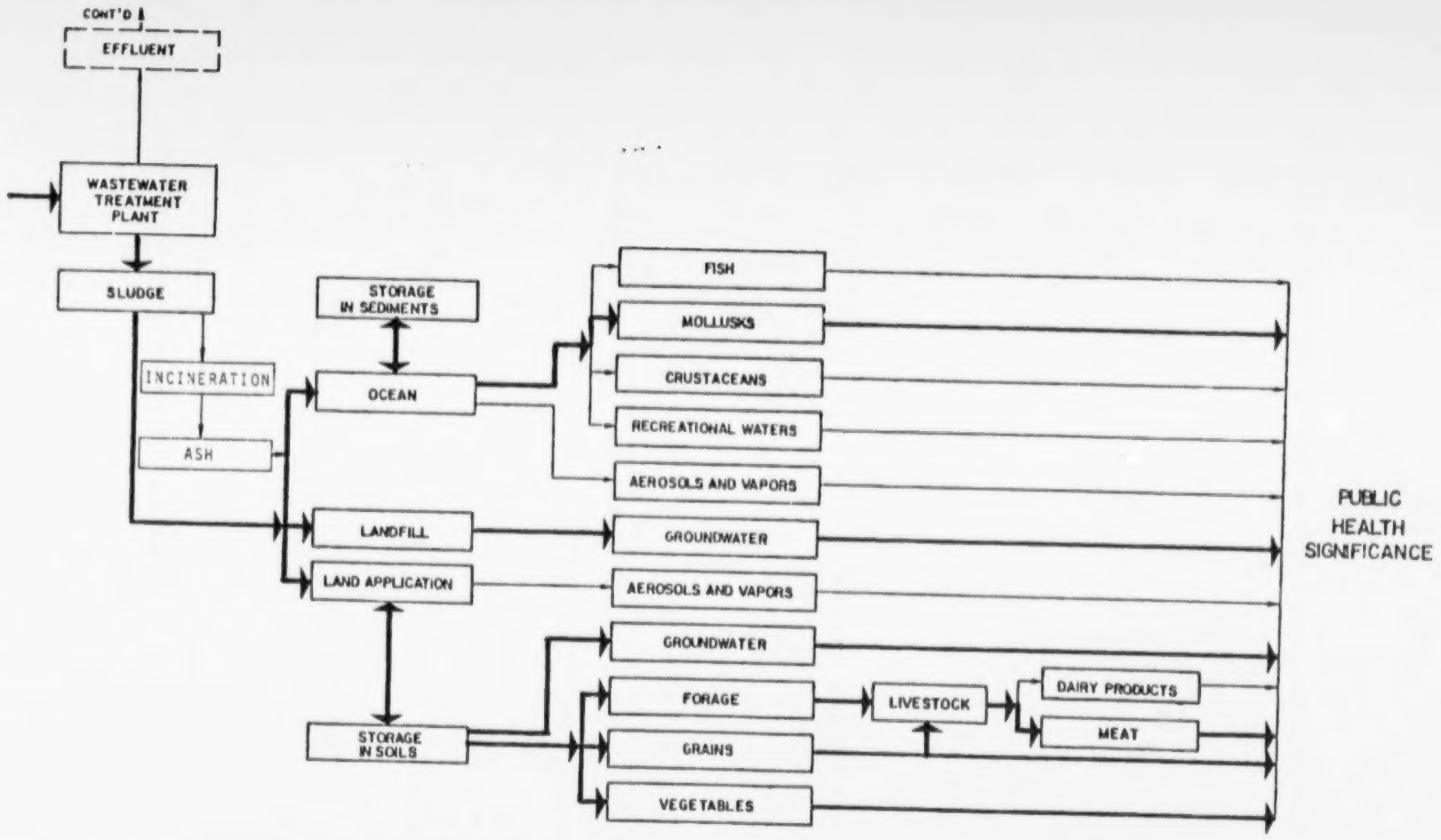


Figure 4. Environmental pathways of cadmium.



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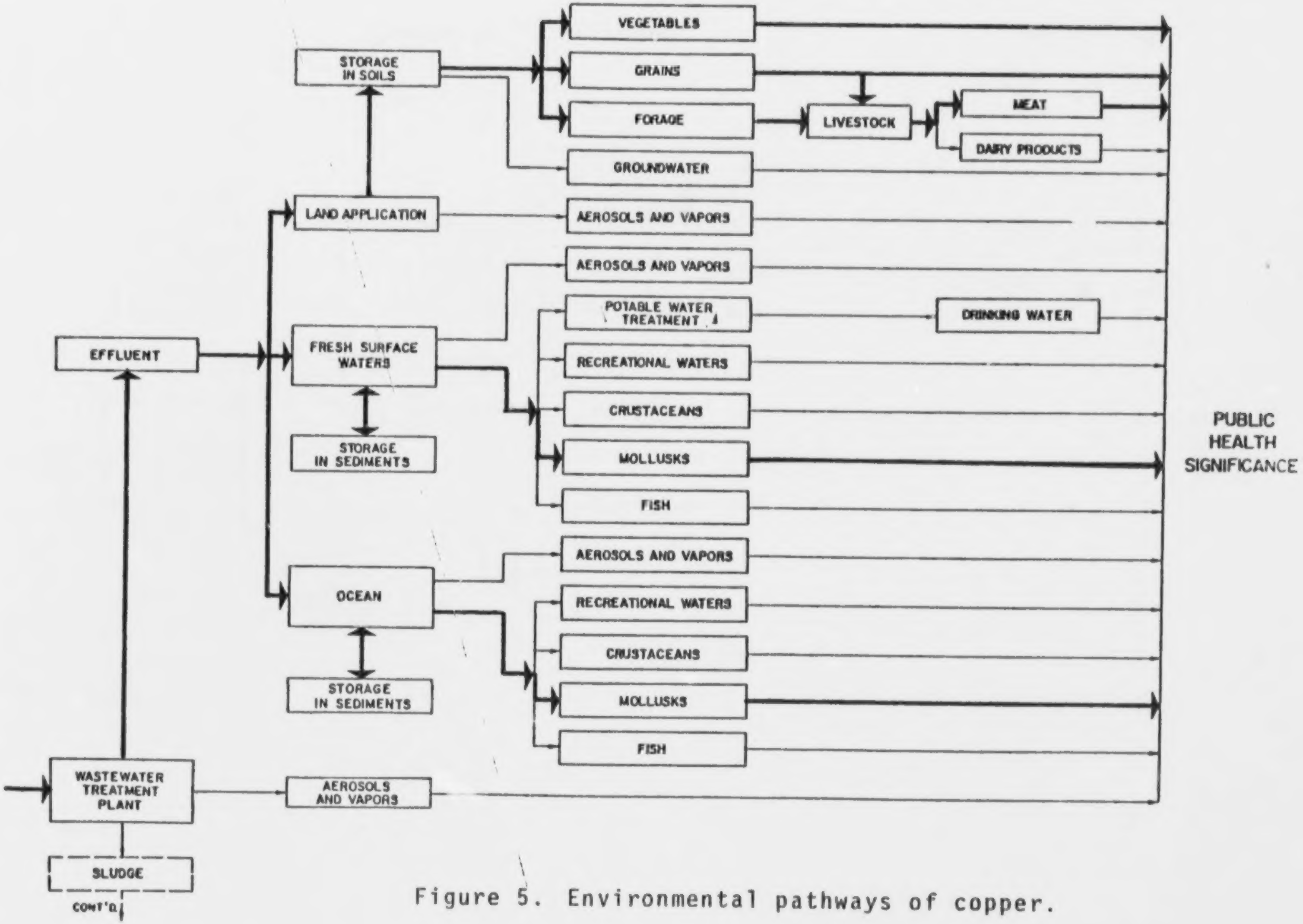


Figure 5. Environmental pathways of copper.

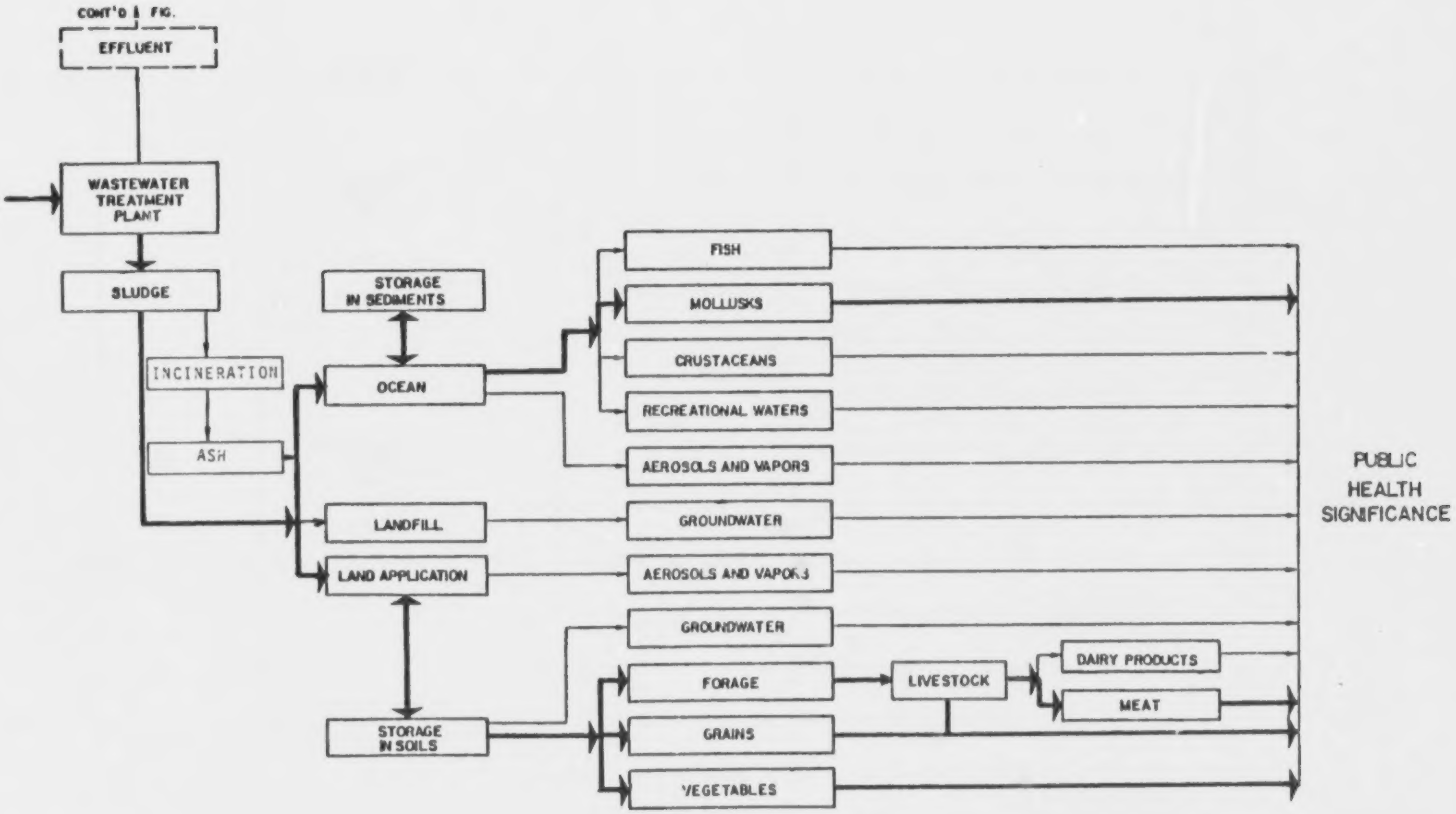


Figure 5. (continued)

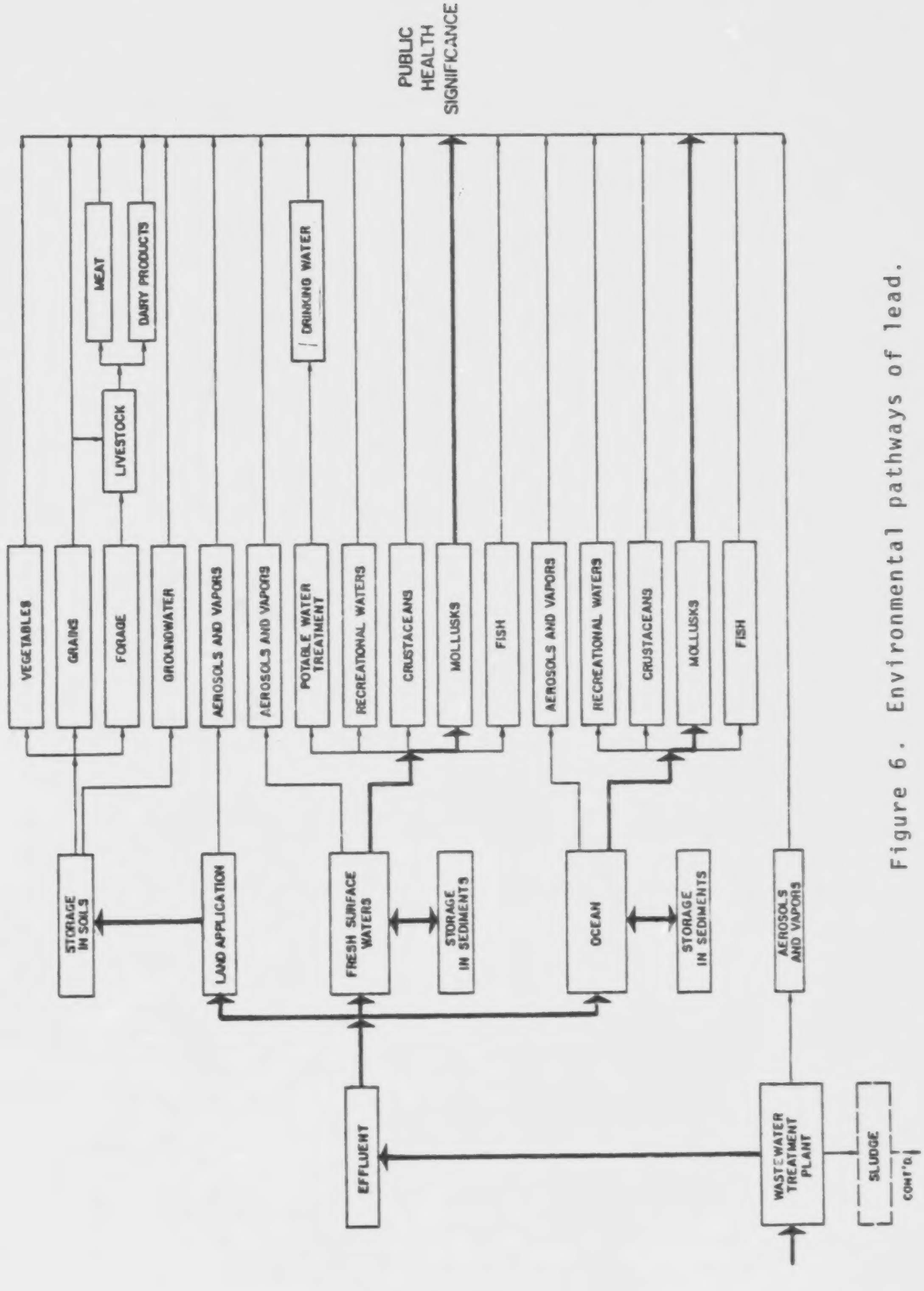
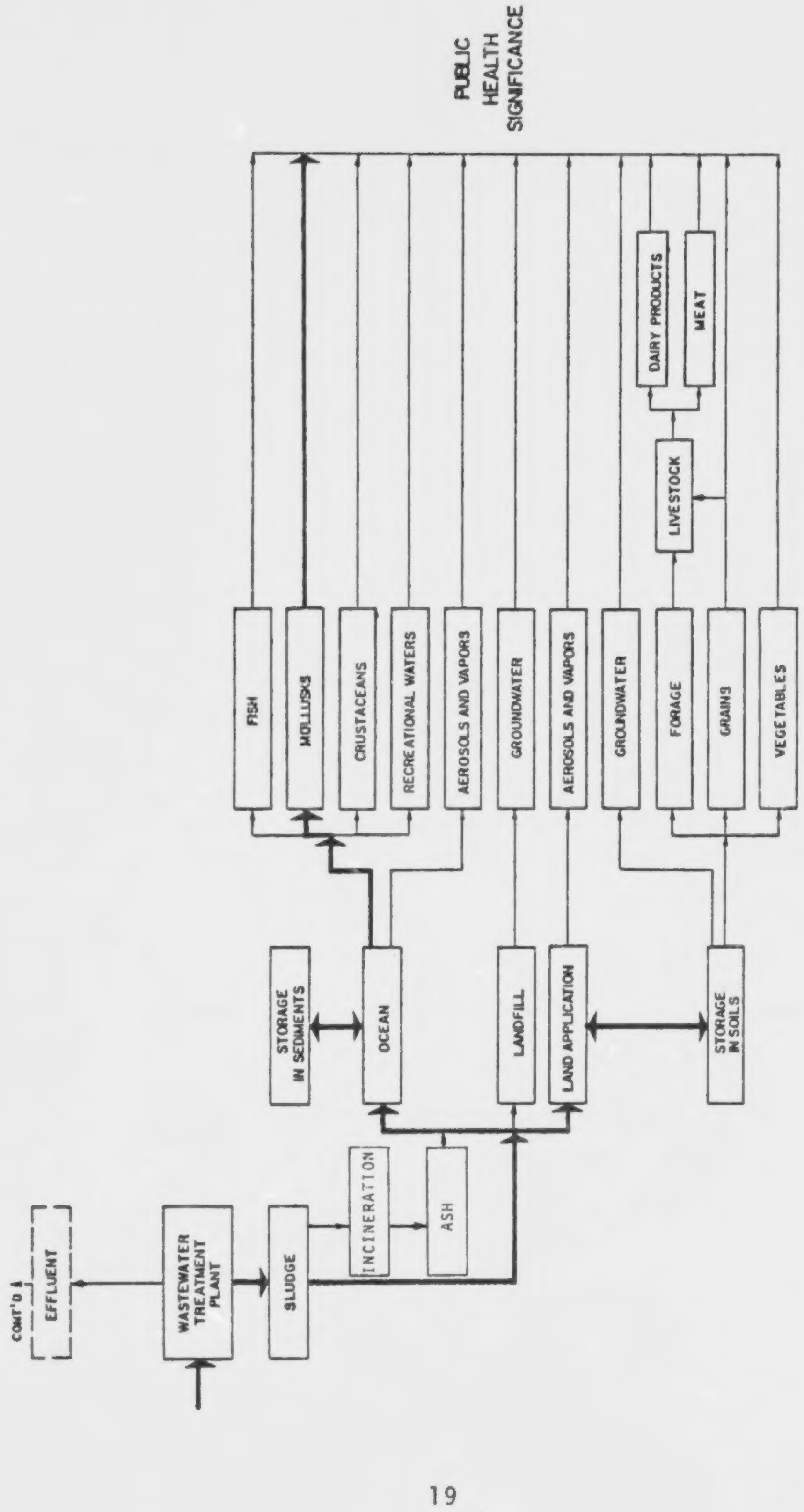


Figure 6. Environmental pathways of lead.



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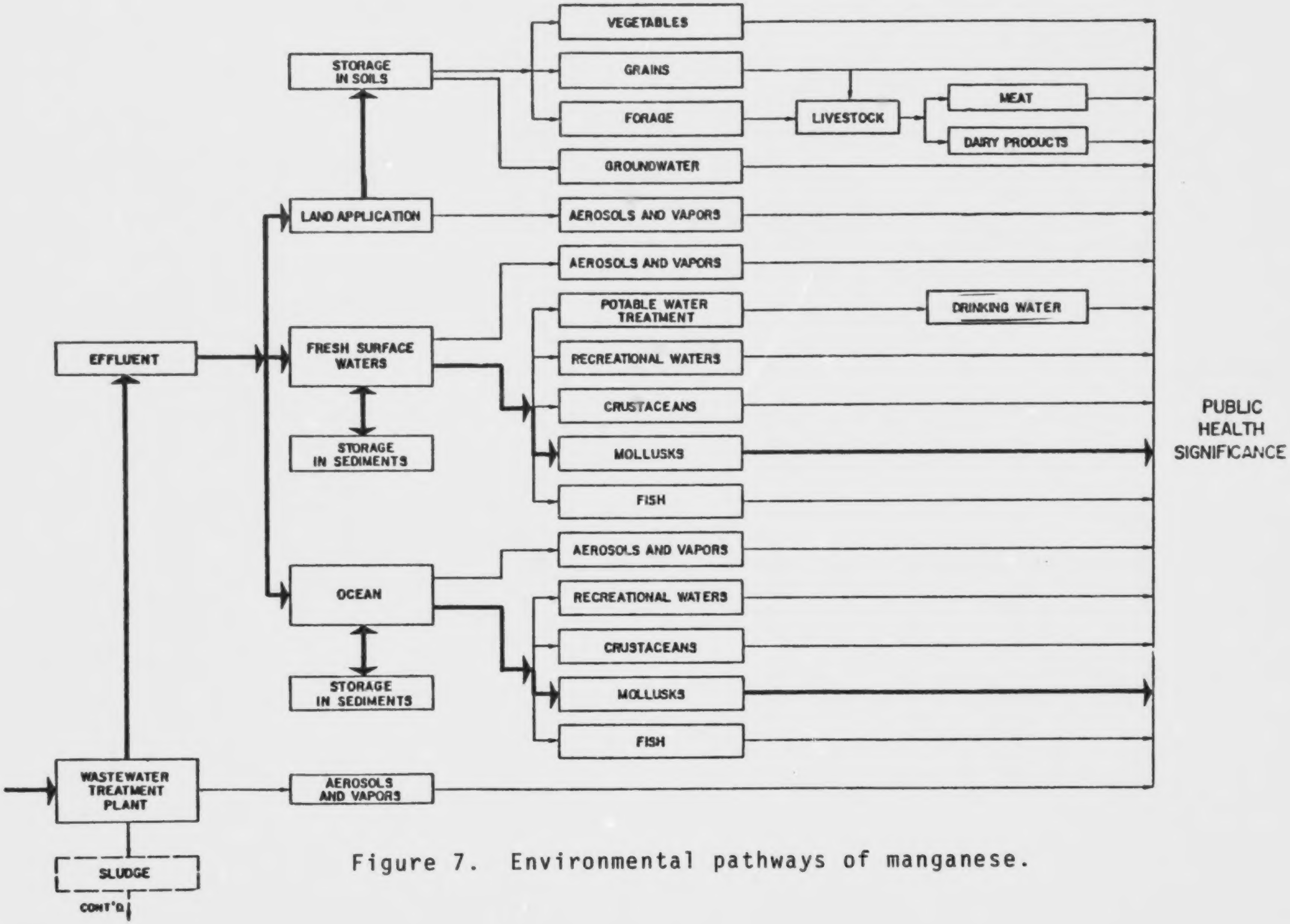


Figure 7. Environmental pathways of manganese.

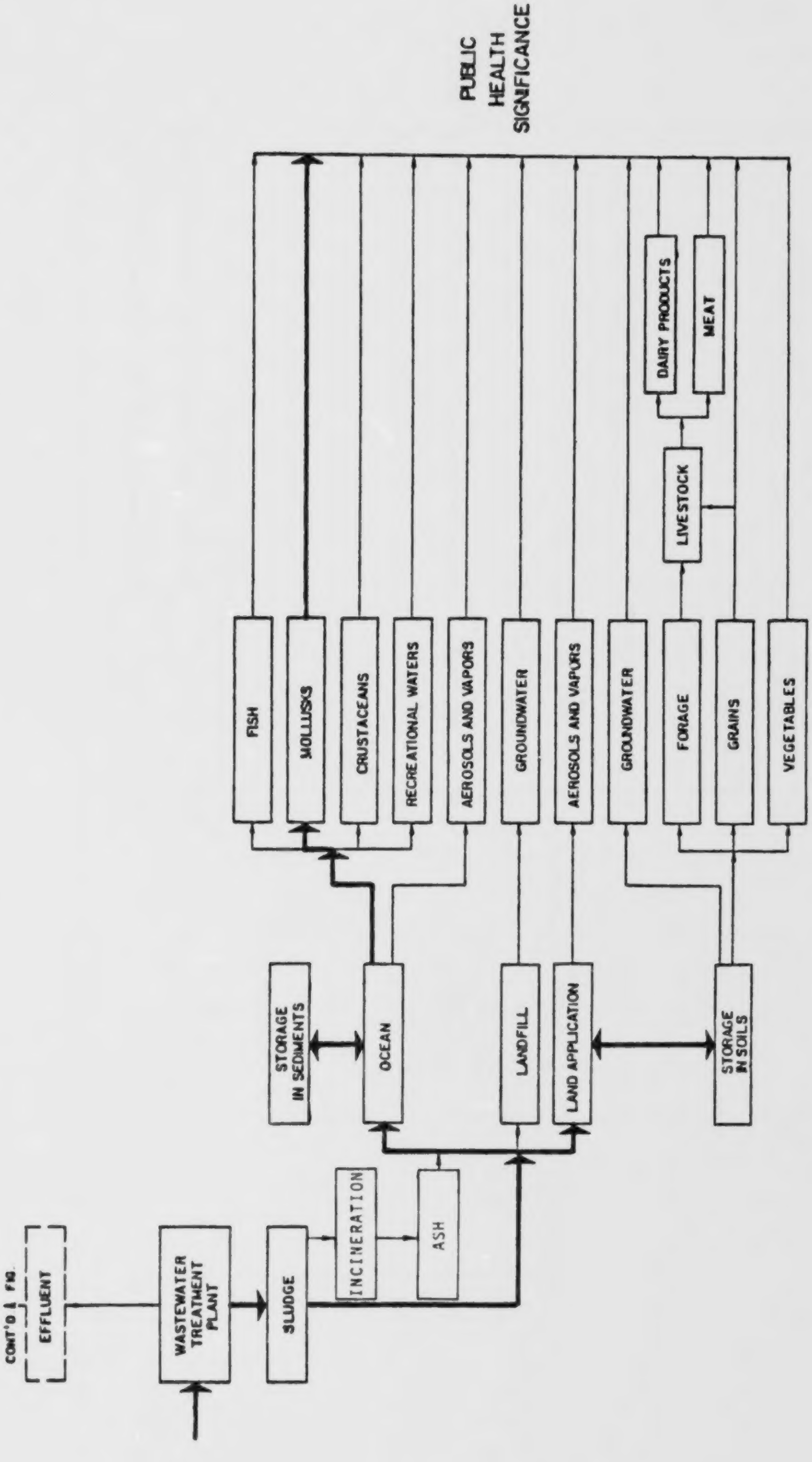


Figure 7. (continued)

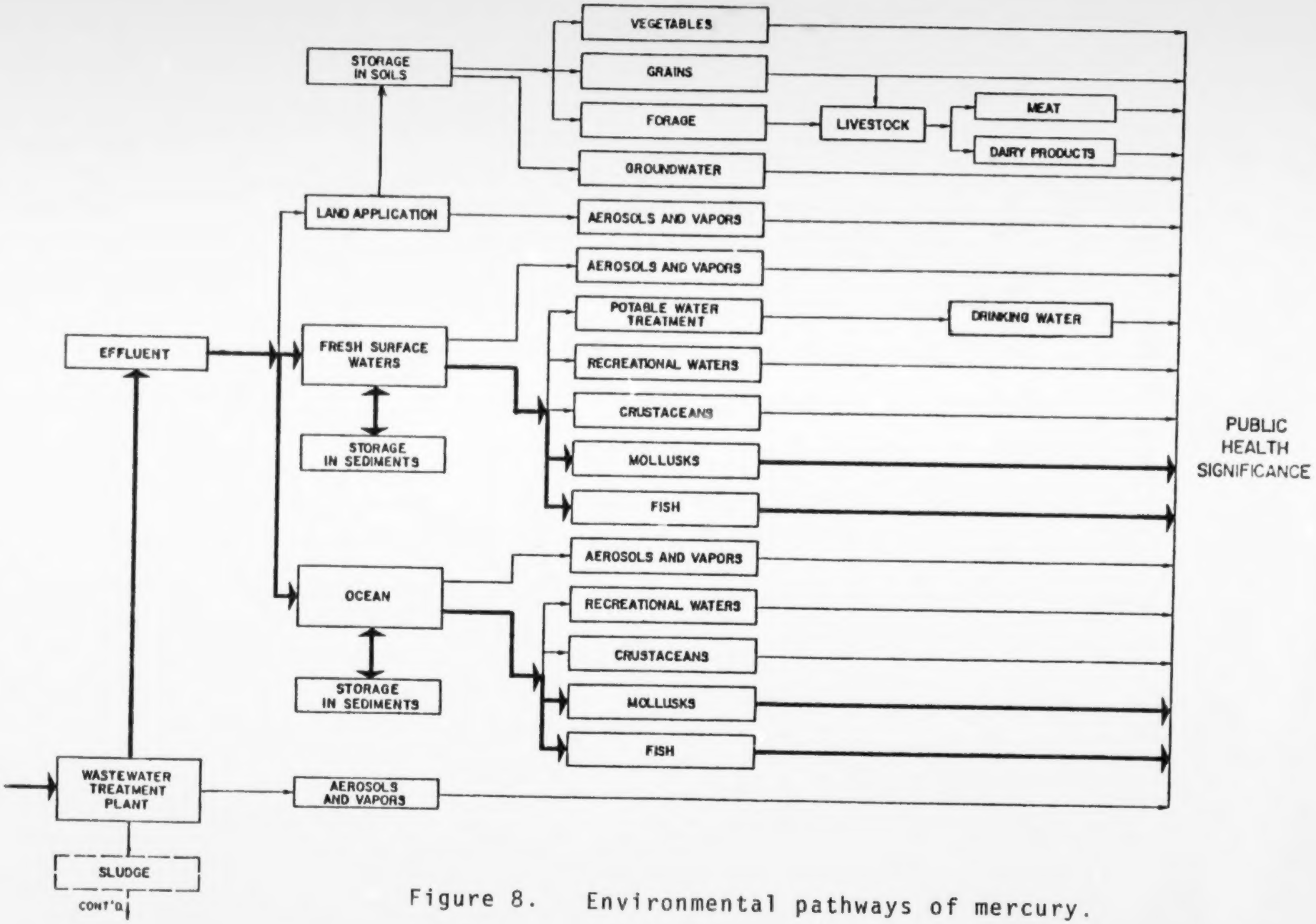


Figure 8. Environmental pathways of mercury.

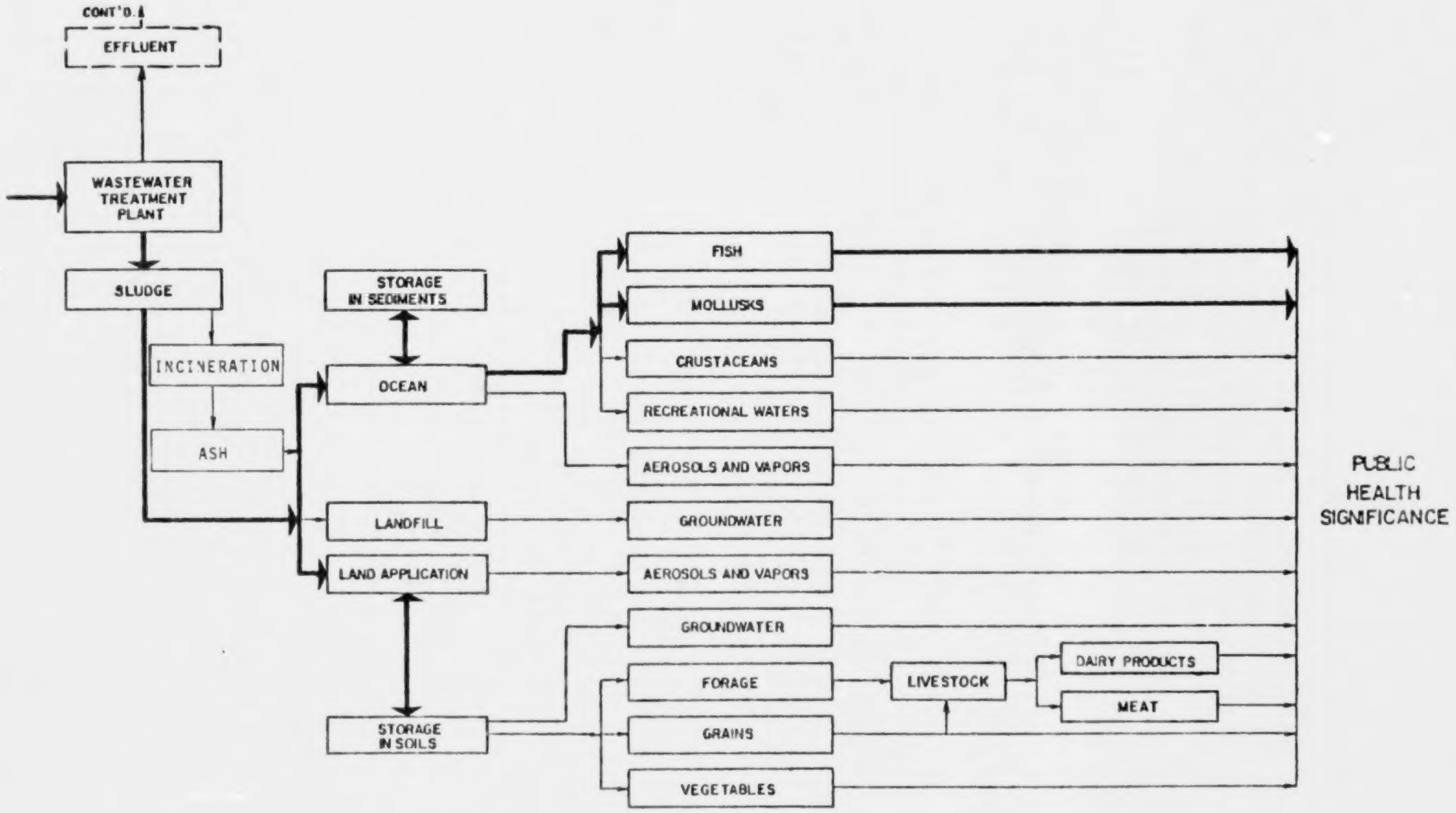


Figure 8 . (continued)

water and saline water may impact public health through the uptake and biomagnification of elemental contaminants by aquatic organisms. The exception to this is arsenic, which is not concentrated by aquatic organisms. The other contaminants, however, may be significantly concentrated, particularly by mollusks. In addition, mercury may be concentrated by fish. When fresh-water bodies containing municipal effluents are utilized for drinking water sources, the water will contain trace quantities of the elemental contaminants. Chemical treatment of water supplies with lime, alum, etc., removes a high precentage of heavy metals. Only under "worst case" conditions could there be a public health problem, but questions remain concerning the chronic health effects to the public of exposures to trace quantities.

When effluents and sludges are applied to land, the major potential impact from elemental contaminants (particularly cadmium, copper, and manganese for the environmental pathways examined) appears to be uptake by some vegetables, particularly leafy types. In addition, cadmium and copper may be taken up by grains and forage. These contaminants subsequently may be transported back to man through meat products from livestock that feed upon the grains and forage.

Throughout the literature review, there was a lack of information concerning the chemical form of the elemental contaminants. Most of the elements do not exist in the ionic form but are instead complexed with a myriad of organic and inorganic ligands and complexing agents. The chemical form will dictate, to a large extent, behavior of the elemental contaminants in various environmental situations, including the human body. Most of the literature examined, however, deals only with total concentrations of the elemental contaminants (resulting from analysis after acid digestion), without regard to chemical form. Thus, there is a need to determine the chemical states of the elemental contaminants and their consequent effect on public health.

Although some research has been performed on removal of metals by secondary biological treatment processes (activated sludge, trickling filter, lagoons), the effect of process variables (lb O_2/lb BOD, sludge wastage rate, mean cell residence time, etc.) upon the chemical states of individual metals and their subsequent removal is not well documented. This information is needed so that conventional secondary plants can be operated to obtain maximum removal of specific metals. Metal removal by chemical coagulation needs further investigation to optimize process variables and maximize removals. The relatively low cost of this tertiary application makes it attractive as a technique to aid in heavy metal control, although a metal-rich sludge is produced.

Research is needed into the chemical state of the metal as it leaves the treatment plant and the effect of this state on the metal's fate in land disposal systems. Some research has been conducted into the removals afforded by migration through various soil systems to which effluent or sludge has been applied. However, further theoretical study is required into the mechanism of removal, especially the factors affecting the adsorption of metals by soil particles, the effect of operating variables on removals, competition between metals and other contaminants for adsorption sites, and desorption phenomena. In addition, the long-term capacity of various soils to adsorb metals needs to be studied.

Another area requiring research concerns the use of effluent or sludge containing heavy metals for crop irrigation and fertilization. It is known that soils will concentrate most heavy metals applied in wastewater and sludge within the top few inches of the soil horizon. It has also been found that crops grown in these soils will uptake higher concentrations of these metals. However, little literature has been uncovered that documents the potential health hazards to man and domestic animals of consuming elevated metal concentrations in food.

There is only one subject area recommended for further research regarding heavy metals discharged from municipal treatment plants to surface water systems: shellfish ingestion and concentration. Some information in this area is being provided by technology transfer from nuclear research, which is evaluating the concentration and storage of radionuclides in shellfish.

As was previously mentioned, an important future research area is the chronic health effects resulting from various chemical states of the elemental contaminants. Also of vital importance is the synergistic effects of various elemental contaminants and their species.

BIOCIDAL CONTAMINANTS

The biocidal contaminants considered in this study are as follows:

- DDT
- DDD
- DDE
- Aldrin
- Dieldrin
- Endrin
- Chlorinated hydrocarbons
- Arsenated hydrocarbons
- Organonitrogen pesticides
- Organophosphorus pesticides
- Herbicides
- Soil sterilants

The literature review did not expose any significant, nationwide problem with biocides in wastewater effluent or sludge. There are specific local problems associated with the discharge of biocidal manufacturing facilities, but these problems do not require a national effort beyond alerting local authorities to a particular situation.

When considering the large amounts of diverse biocides produced and utilized in agriculture as well as other pursuits, it becomes clear that most biocides enter the environment by pathways other than sanitary waste systems.

In determining the critical pathways of biocidal contaminants, the following were selected for examination:

- DDT
- Dieldrin
- 2, 4-D
- Parathion

Concentrations of 2, 4-D and parathion are so low in the input to wastewater treatment plants that their pathways through the environment become negligible. The environmental pathways of DDT and dieldrin are presented on Figures 9 and 10, respectively. As revealed by examining the environmental pathways, the mass loading of the selected pesticides usually ends up either in the soil matrix or in the sediments, depending on the disposal mode. Both aerosolization and uptake by aquatic organisms of the pesticides may be, at times, a problem. However, as previously discussed, the pesticide input to these facets of the biosphere from municipal wastewater systems is negligible compared to other sources.

SYNTHETIC/ORGANIC CONTAMINANTS

There are three major sources of synthetic/organic chemicals in municipal wastewater effluents:

- Industrial chemicals contained in industrial wastes discharged into municipal collection systems
- Naturally occurring organic chemicals and metabolic products generated by the human body and by bacterial activity
- Chemical transformations which are caused by wastewater treatment processes such as chlorination, chemical treatment, heat and pressure treatment, etc.

The literature reviewed indicates that little is known about the synthetic/organic contaminants from any of the sources. Specifically, there is almost a total lack of knowledge about synthetic/organic contaminants in the following basic areas:

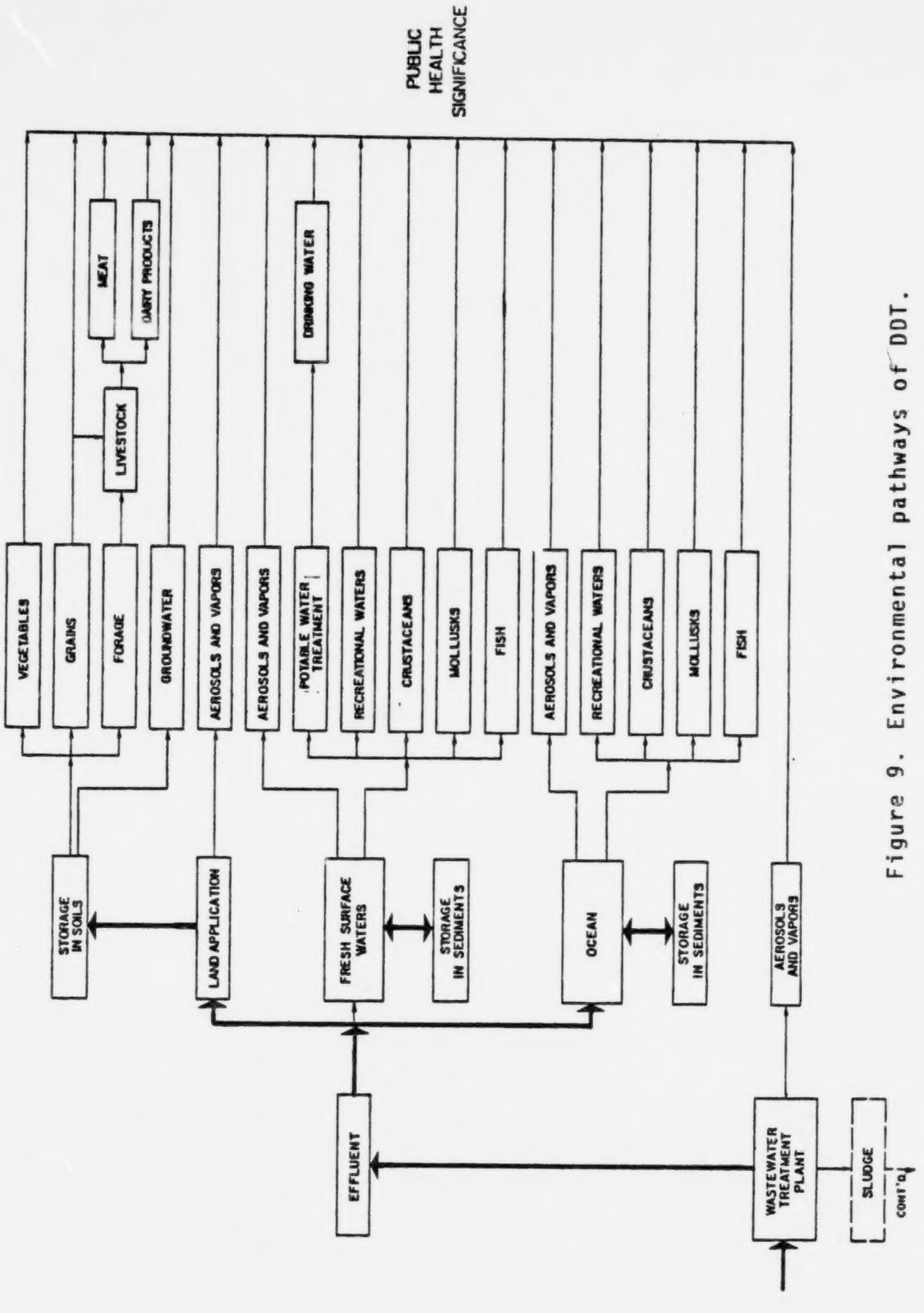


Figure 9. Environmental pathways of DDT.

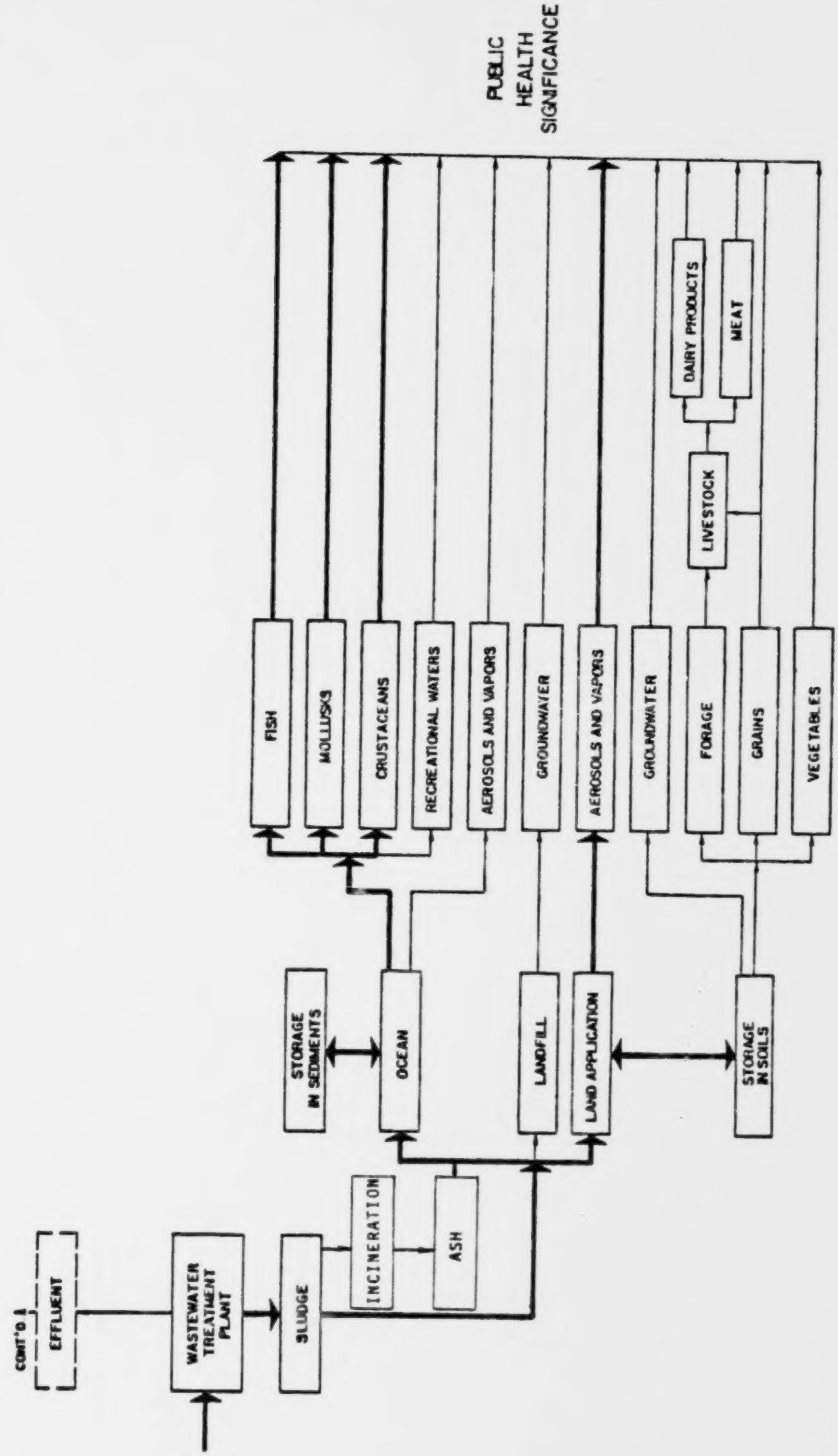


Figure 9. (continued)

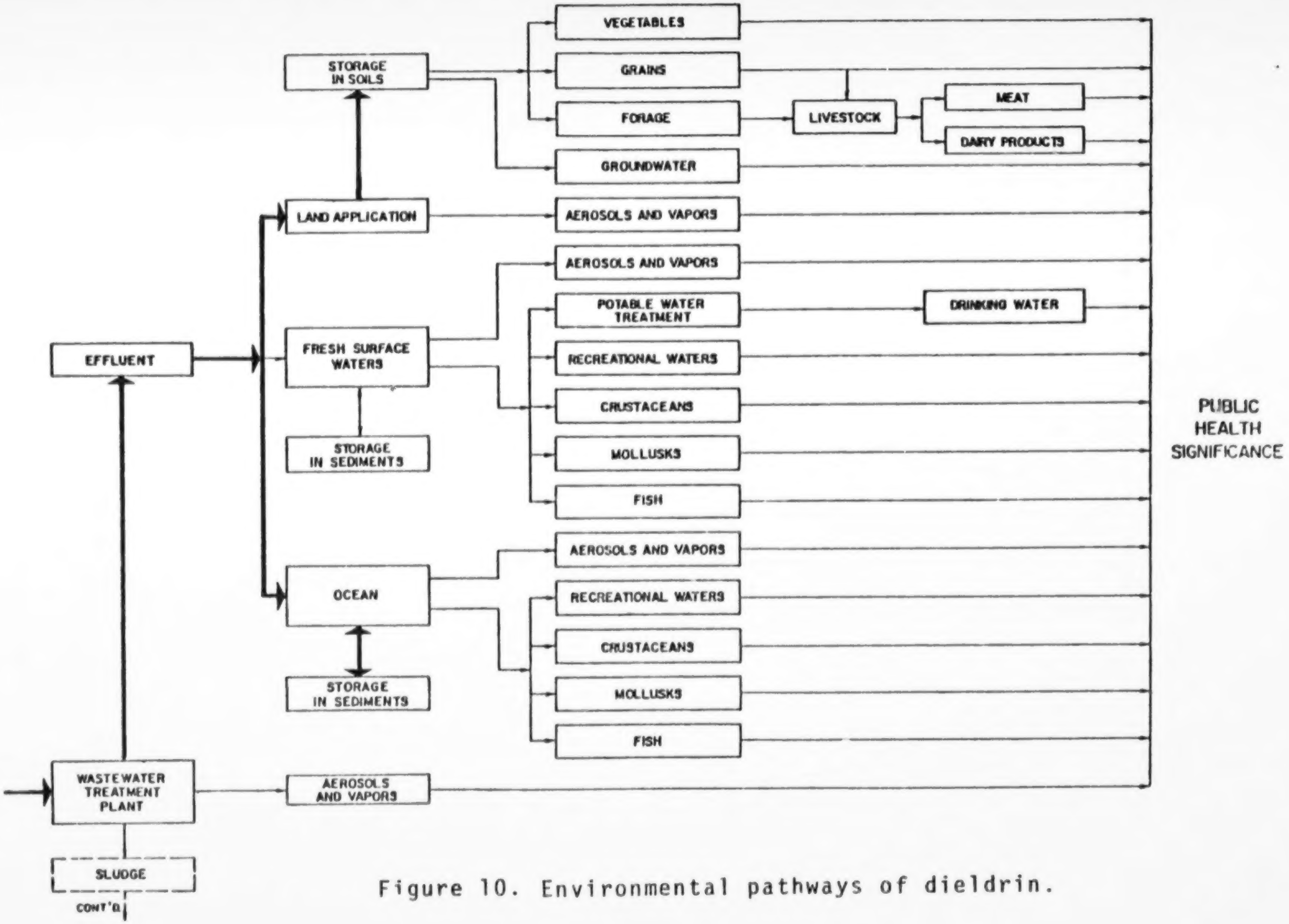


Figure 10. Environmental pathways of dieldrin.

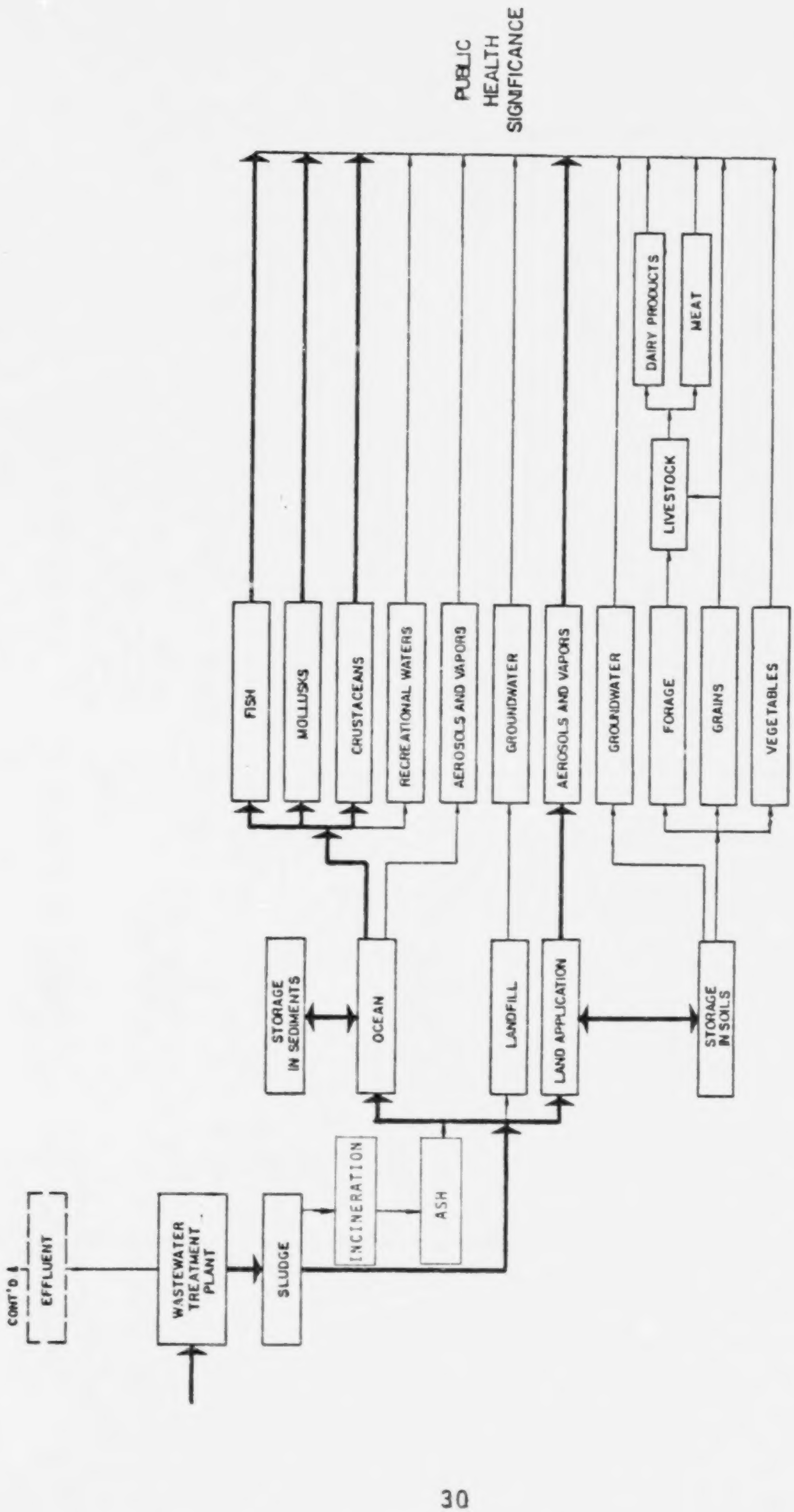


Figure 10. (continued)

- The concentration of these contaminants in raw sewage or treated wastewaters

- The concentration of these contaminants in wastewater sludges

- Methods of analyzing many of the organic chemicals easily, accurately, and inexpensively

- Health effects of chronic and acute dosages to humans.

To further complicate the problem, little is known about the fate of synthetic/organic contaminants in various environmental situations. The environmental pathways of two synthetic/organic contaminants, PCB's and chloroform, are shown in Figures 11 and 12, respectively. The mass balances developed for PCB's indicate that input concentrations to municipal wastewater treatment plants are extremely low and that there is little discernible impact on any of the environmental pathways. The majority of PCB's remaining in wastewater effluents after treatment will end up in the sediments or soils, depending upon the final disposal mode. The potential public health problem associated with PCB's in municipal wastewaters occurs when significant raw sewage input is concentrated in wastewater sludges. Subsequent disposal of the sludges can cause public health problems through biomagnification by aquatic organisms if disposal is to the ocean, and there is localized vaporization during land application or groundwater contamination if leaching occurs.

Examination of the environmental pathways of chloroform indicates that due to its low vapor pressure, the potential impact of chloroform to public health occurs through vaporization and transport through the atmospheric pathway. The creation of chloroform during the chlorination process is well established.

It may be speculated that residual trace organics in municipal wastewater effluents can be converted to carcinogenic compounds during chlorination of municipal water supplies that are drawn from effluent-receiving surface water bodies.

The concentrations of synthetic/organic contaminants in municipal wastewater are below levels that will cause acute health effects. However, there is little information concerning the chronic health effects resulting from exposure to trace quantities of the contaminants over long periods.

There is a dearth of information on virtually every facet of synthetic/organic chemical input, creation, behavior in wastewater treatment and disposal systems, and passage through the ecosystem. Because of this lack of information, it is believed that a "total" understanding is presently beyond technical and economic capabilities. It is suggested that top research priority go to the following areas:

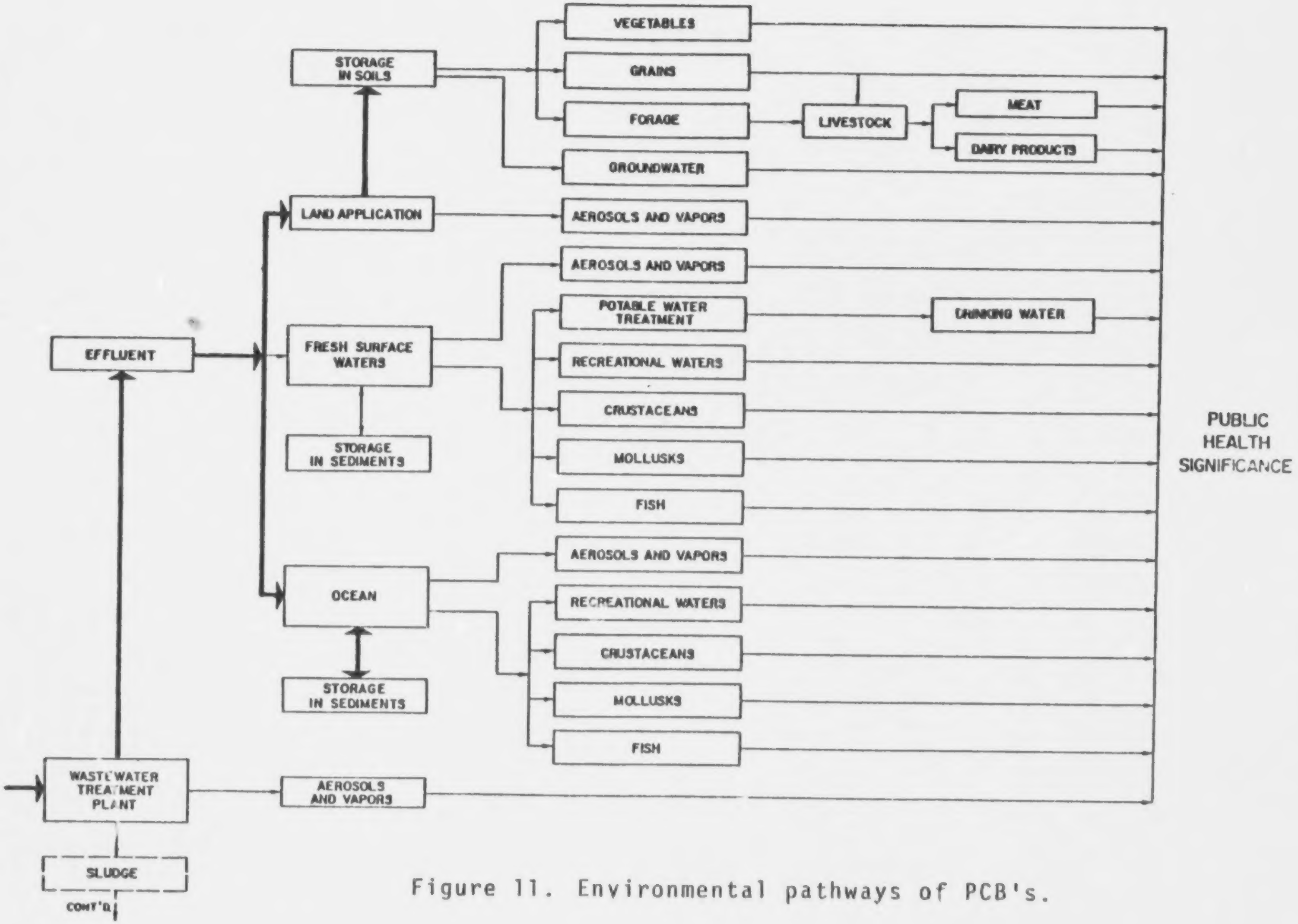


Figure 11. Environmental pathways of PCB's.

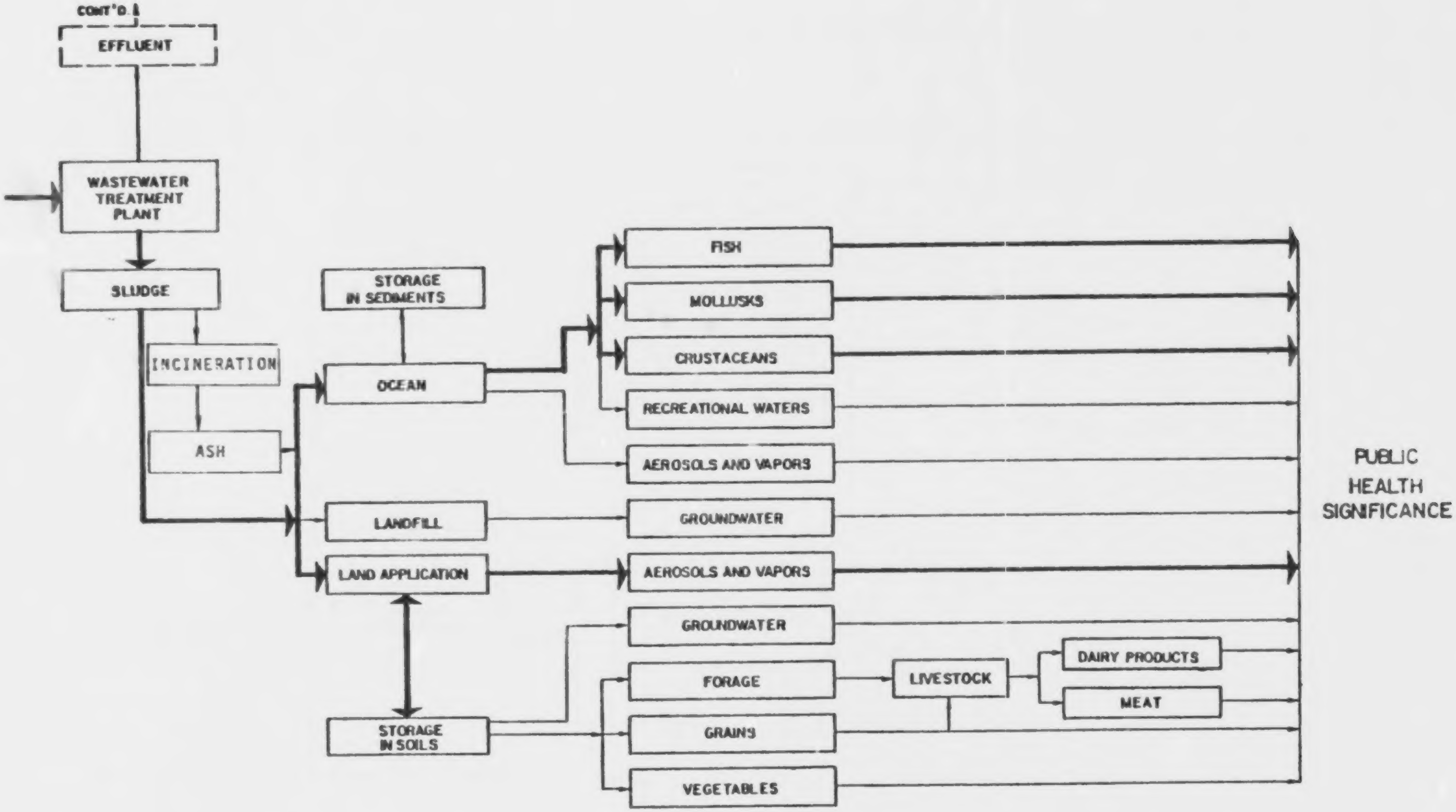


Figure 11. (continued)

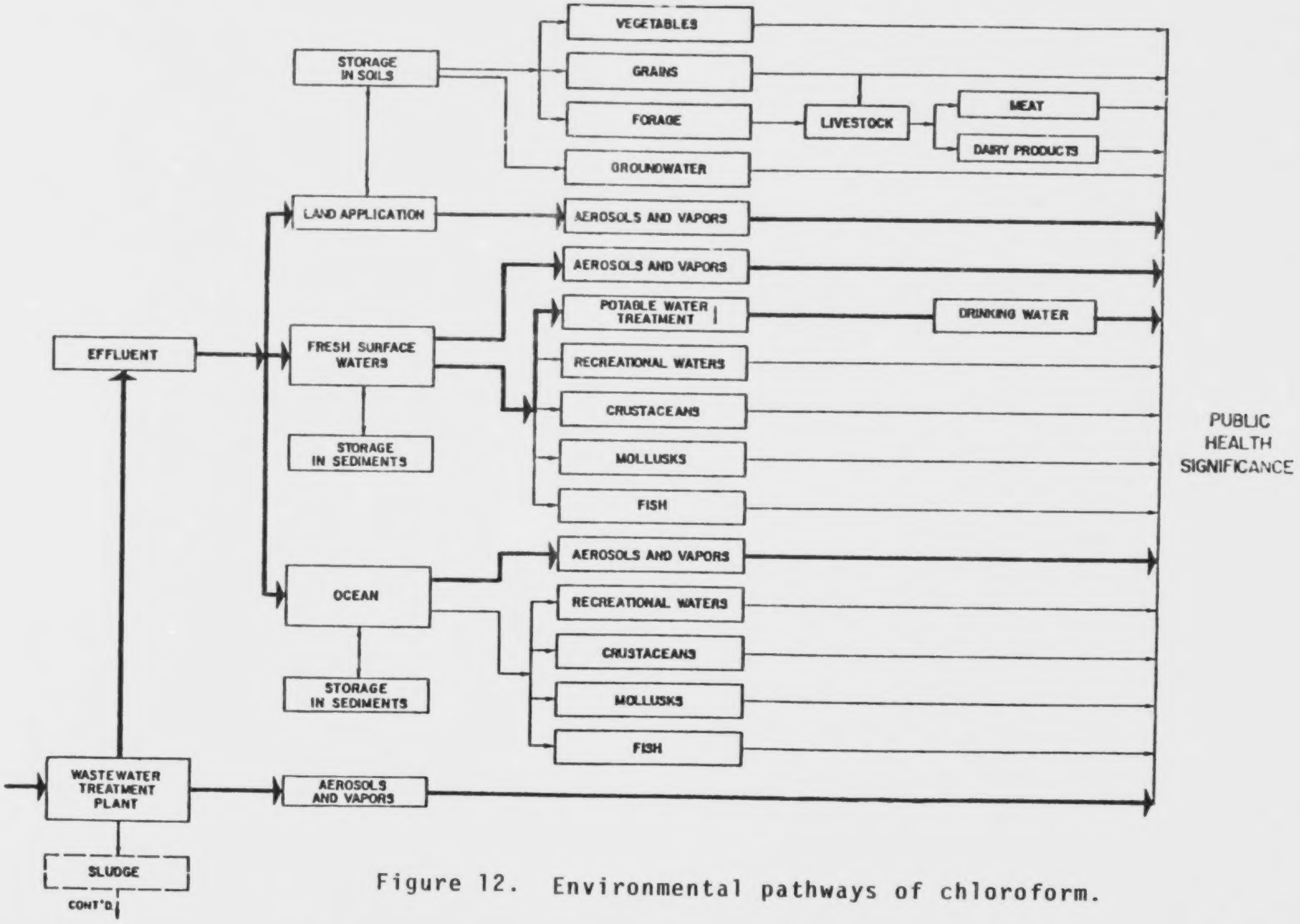


Figure 12. Environmental pathways of chloroform.

- Development of improved analytical techniques in terms of simplicity, accuracy, and cost
- Development, if possible, of an "indicator" chemical or group of chemicals, whose presence implies possible adverse health effects
- Chemical forms and conversions of the synthetic/organic chemicals during the various municipal wastewater treatment processes
- Chronic effects to humans of long-term, low level exposure to synthetic/organic chemicals
- Chemical forms and conversions of the synthetic/organic chemicals inside the human body
- Synergistic health effects of the synthetic/organic chemicals between each other and other chemical contaminants.

BIOLOGICAL CONTAMINANTS

Traditionally, biological contaminants have received the most attention in wastewater treatment, since these contaminants may directly cause public infection. The biological contaminants considered in this study are as follows:

Adenovirus
Coxsackie virus
ECHO virus
Hepatitis virus
Polio virus
Coliforms
Salmonella
Shigella
Leptospirosis
Escherichia coli
Francisella tularensis
Vibrio cholerae
Mycobacterium
Streptococci
Staphylococci aureus
Clostridium botulinum
Clostridium perfringens
Listeria monocytogens
Erysipelothrix rhusiopathiae
Yeasts
Protozoa
Parasitic worms

The effects of wastewater treatment on pathogenic bacteria and their consequent behavior in the environment have generally been well documented. There are, however, considerable gaps of knowledge concerning the behavior of viruses during wastewater treatment and in the biosphere.

Environmental pathway charts were developed for seven biological contaminants. By examining the mass balances of the contaminants through the environment, it was found that with modern wastewater and water treatment disinfection techniques, there is relatively little impact on public health from the biological contaminants. Most documented cases of public contamination were traced to malfunctions of the treatment system processes. The environmental pathways of the biological contaminants are presented in the figures listed below:

Contaminant	Figure
Coliforms	13
E. Coli	14
Salmonella	15
Protozoa	16
Parasitic worms	17
Polio virus	18
ECHO virus	19

The environmental pathways provide several generalized conclusions. For the biological contaminants other than protozoa and parasitic worms, the possible critical pathways are as follows:

- Aerosolization during municipal wastewater treatment processes as well as during land application procedures

- Concentration by mollusks in both fresh-water and saline water environments

- Transmission by vegetables after irrigation application of inadequately disinfected effluents.

One critical pathway for parasitic worms and protozoa is transmission by meat products through livestock after wastewaters and sludges are applied to agricultural lands. This pathway can virtually be eliminated by adequate disinfection procedures. Another criticial pathway is aerosols that are created within the wastewater treatment plant. These aerosols have been researched considerably in the past several years; however, much of the research has focused on coliform survival during the transmission of aerosolized particles. Future research should investigate the survival of other bacterial organisms during aerosolized-particle transmission. The results of these studies should be the development of indicator organisms. Coliforms, a traditional indicator of water quality, tend to desiccate rapidly when aerosolized. Therefore, these organisms may not be effective in indicating when aerosolization of biological contaminants poses public health problems.

With regard to the destruction or removal of pathogenic bacteria, protozoa, and parasitic worms within the treatment plant (especially by disinfection), no further research is recommended. Rather, continuous surveillance and monitoring of existing systems are needed to protect public health.

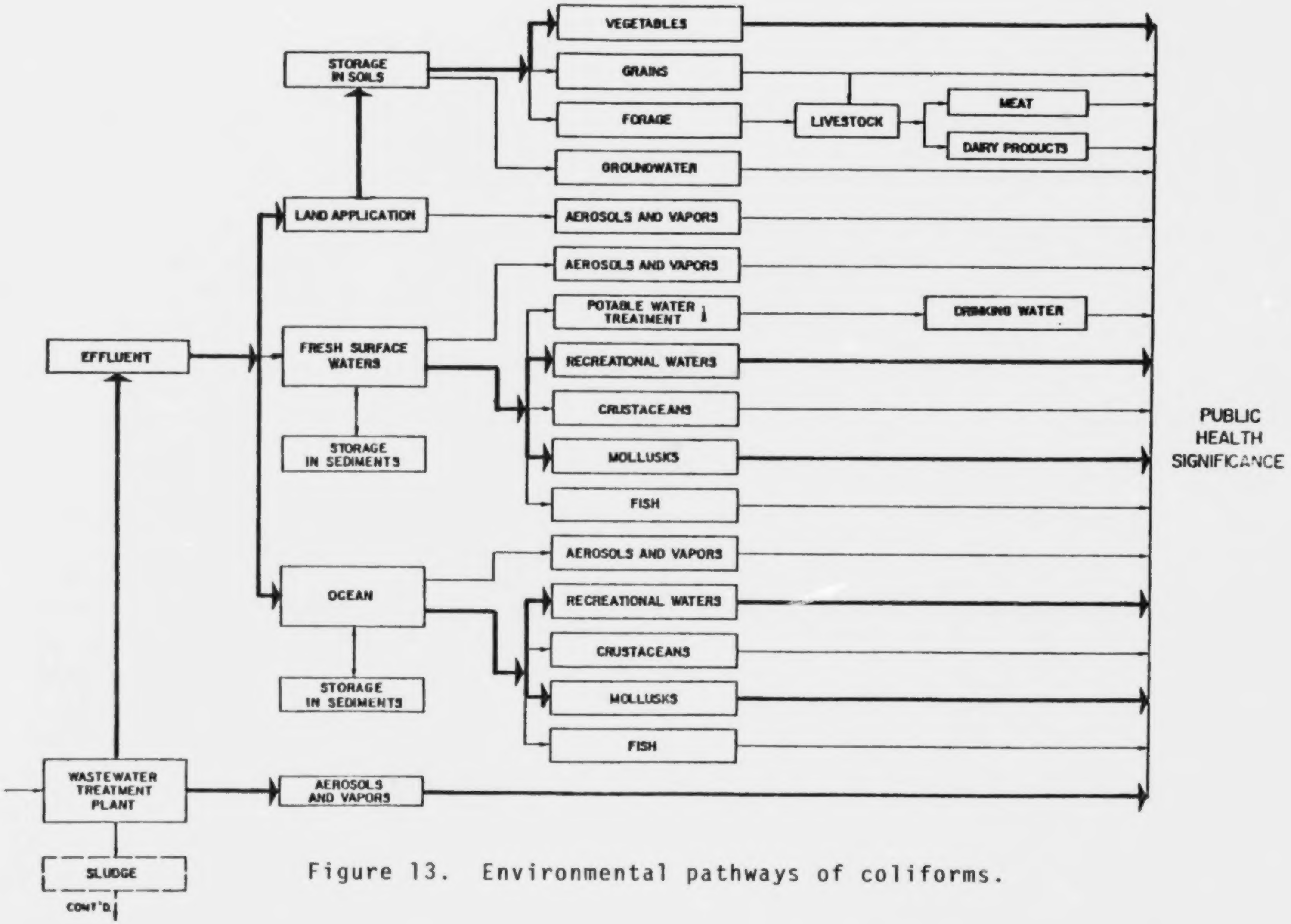
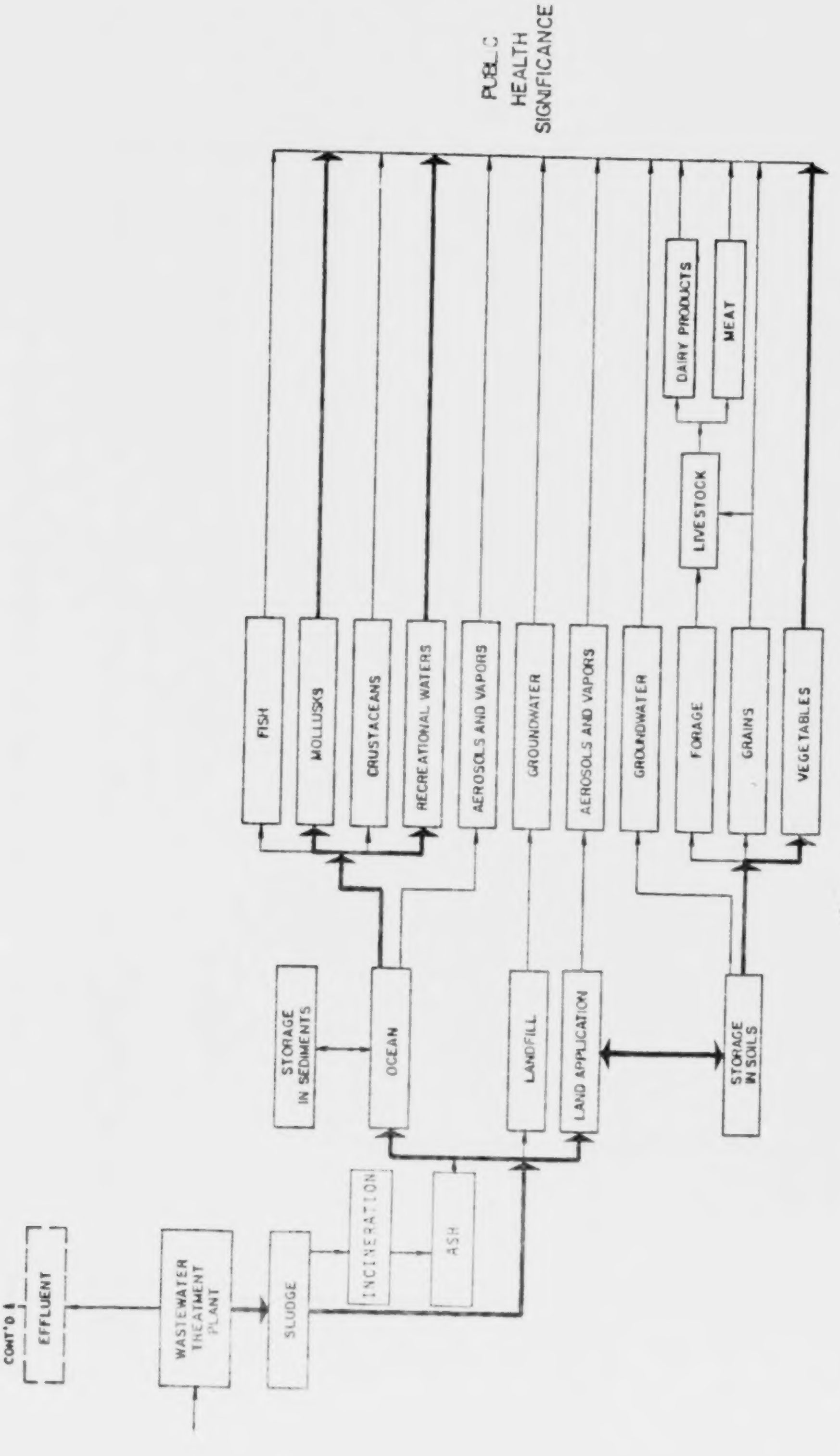


Figure 13. Environmental pathways of coliforms.



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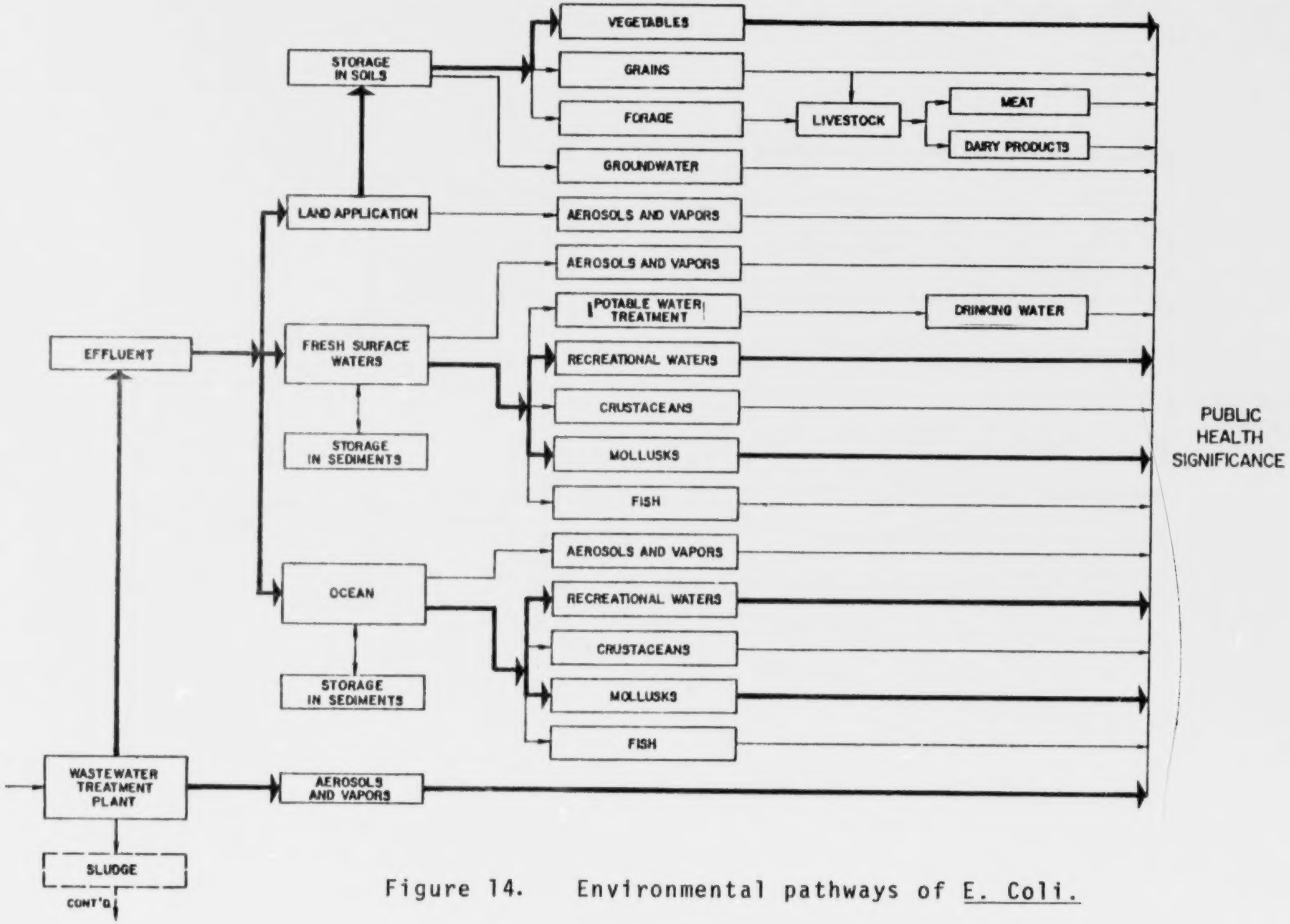
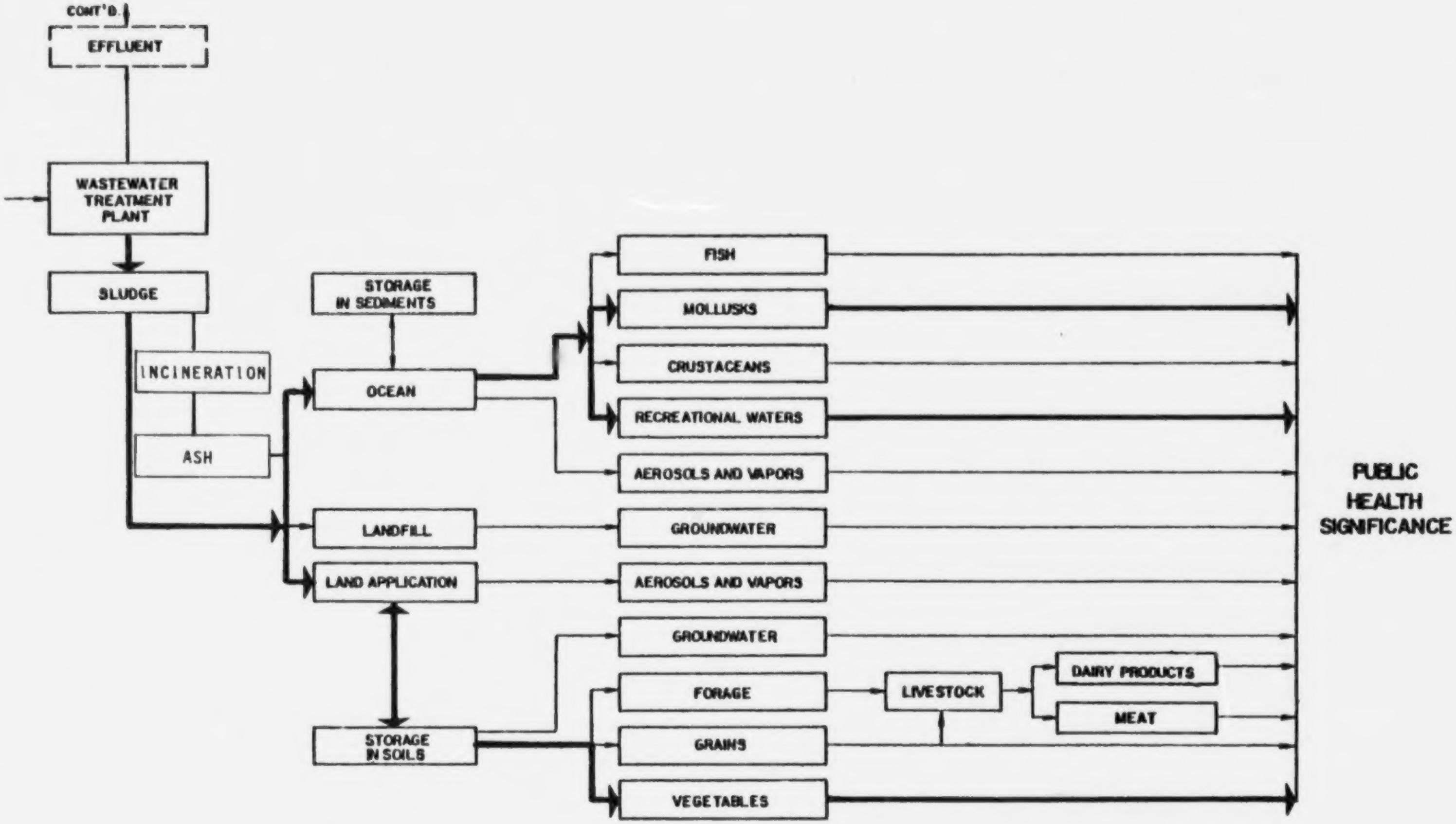


Figure 14. Environmental pathways of E. Coli.



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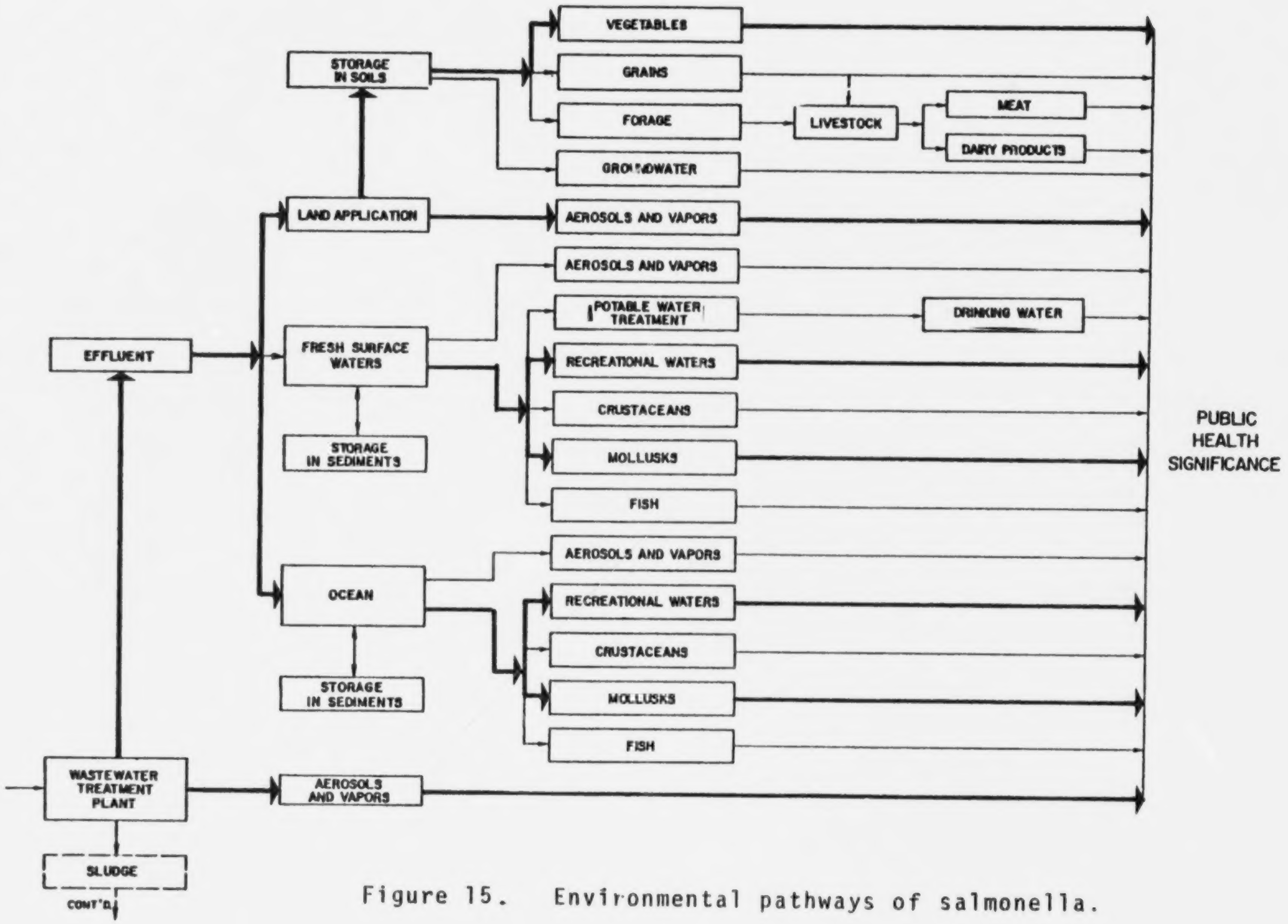
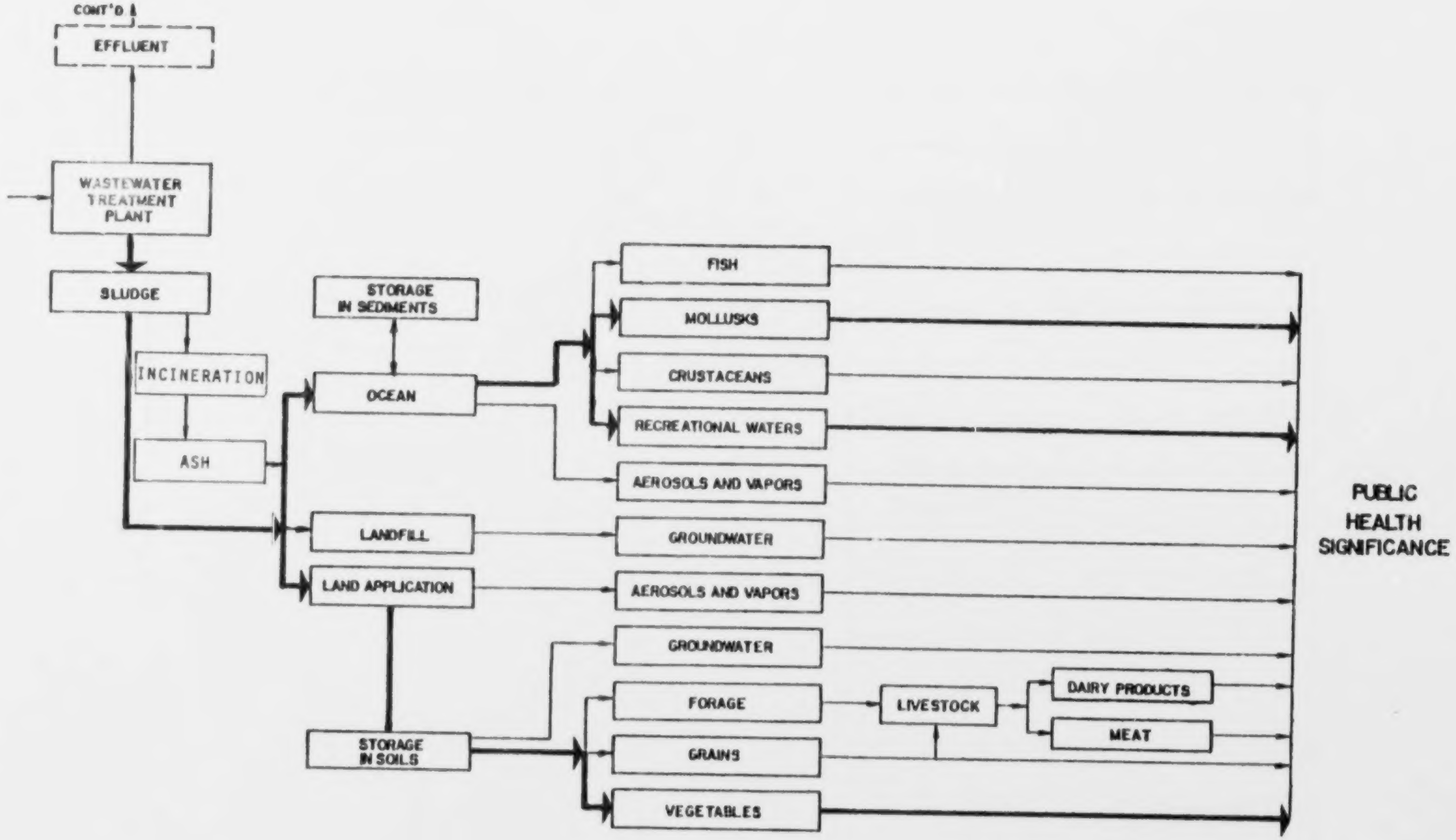


Figure 15. Environmental pathways of salmonella.



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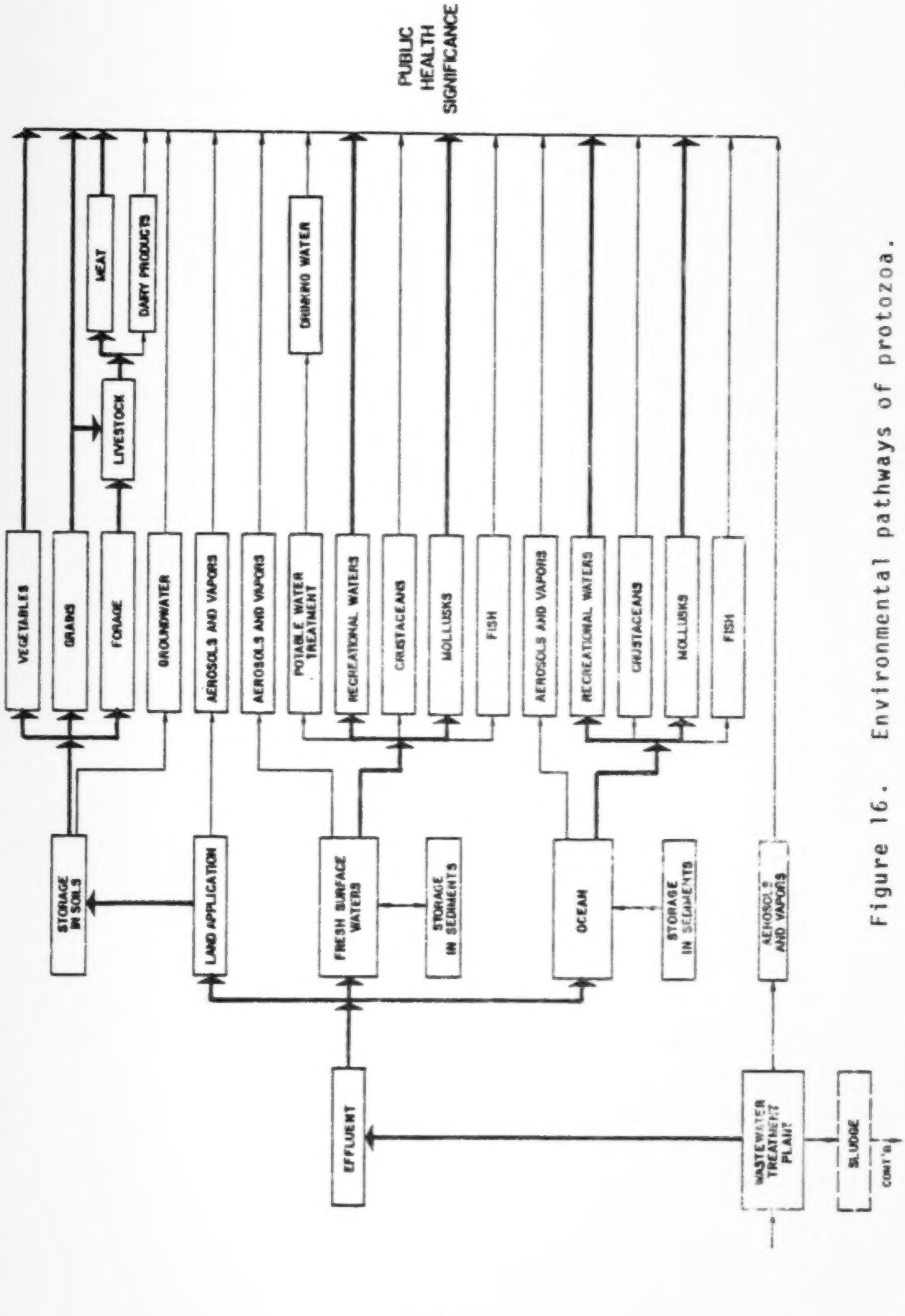
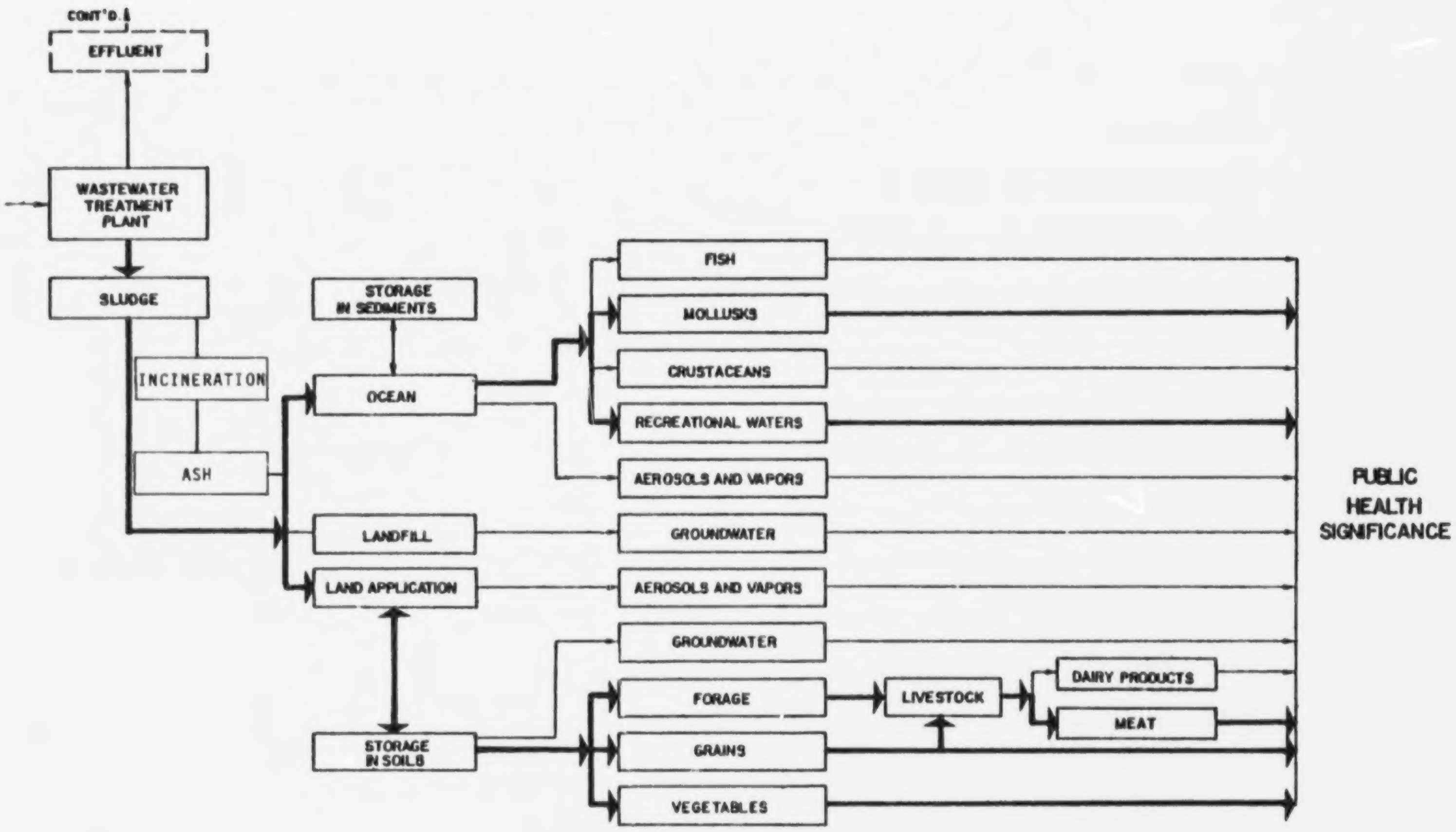


Figure 16. Environmental pathways of protozoa.



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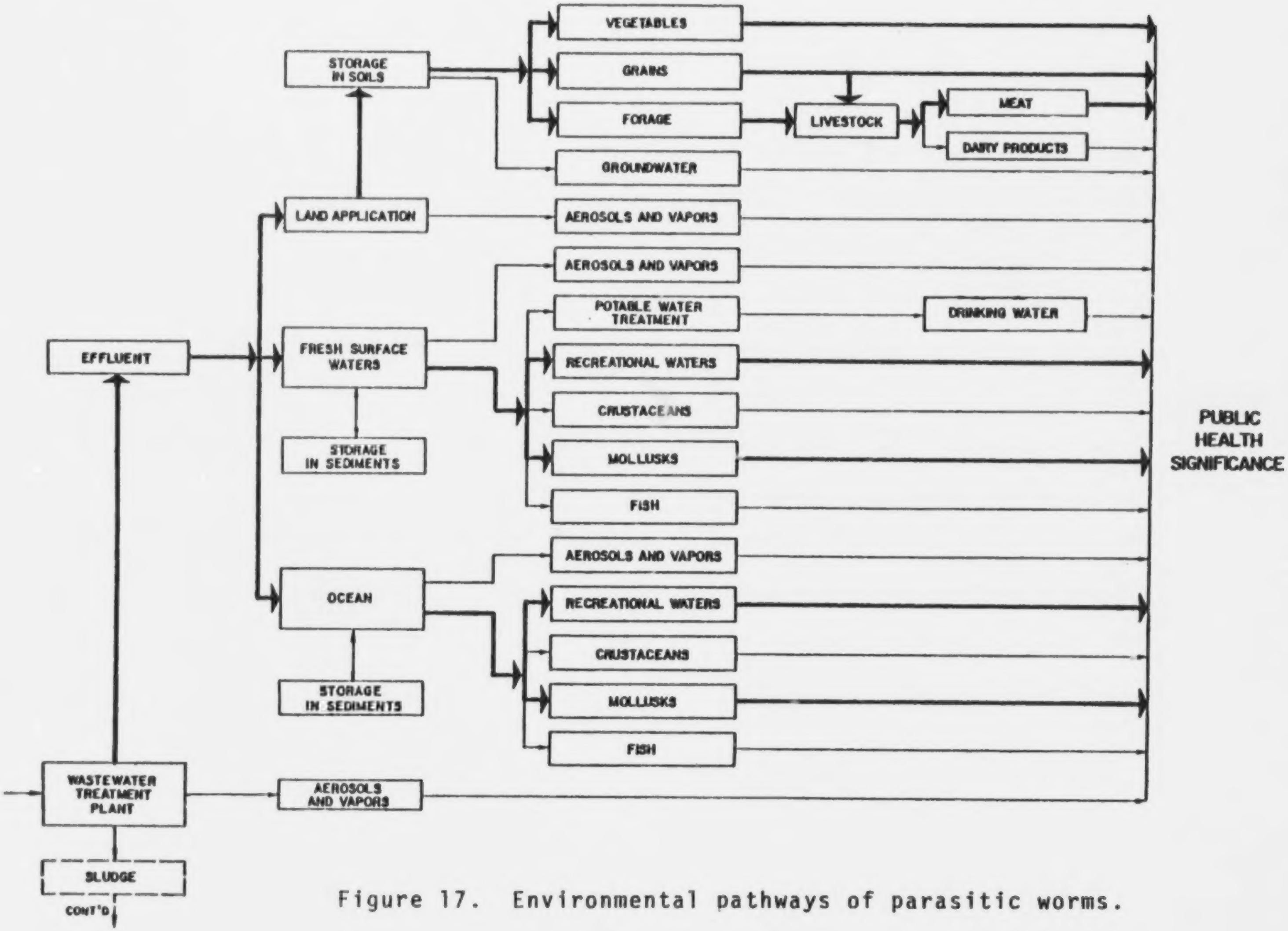
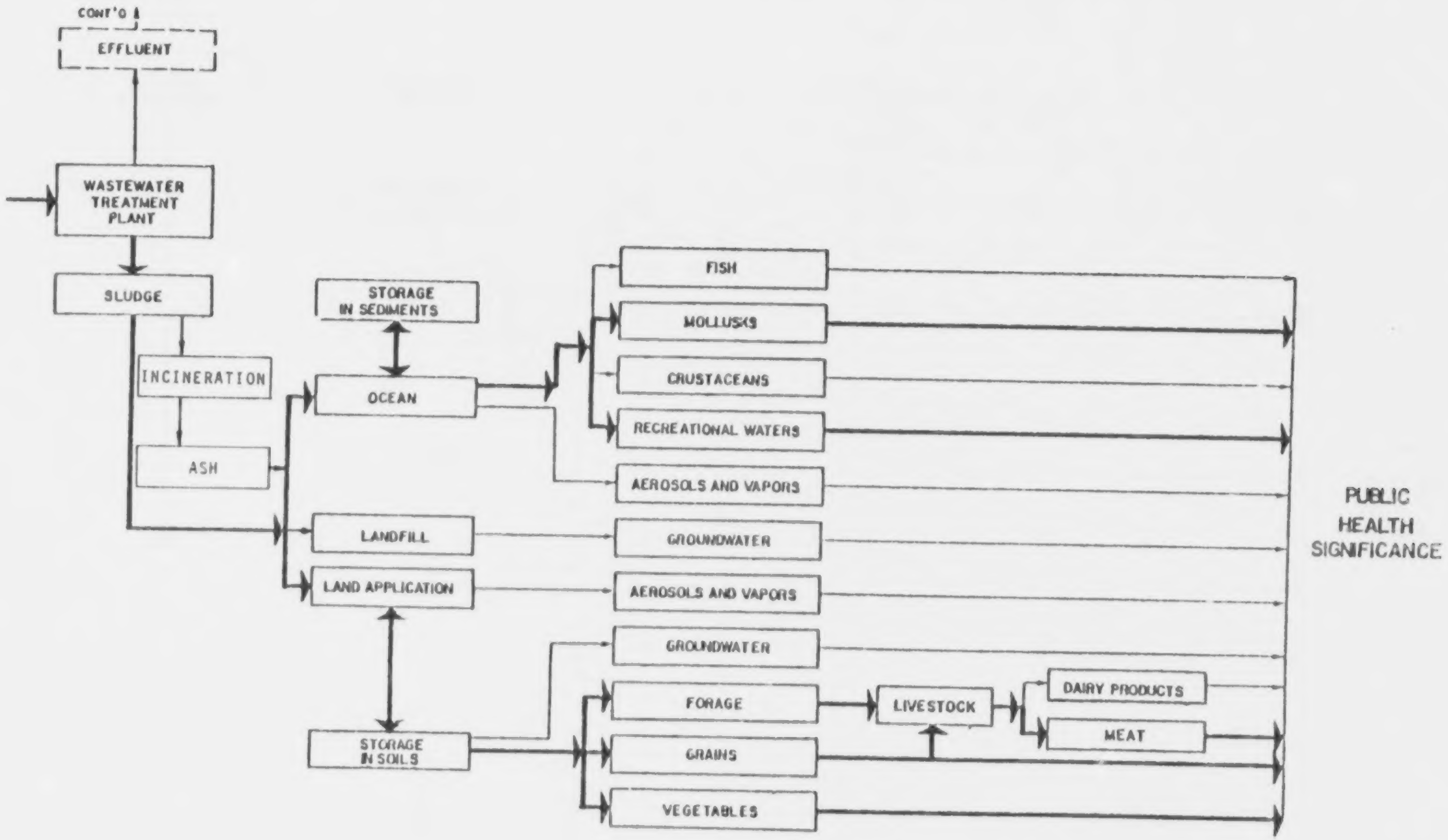


Figure 17. Environmental pathways of parasitic worms.



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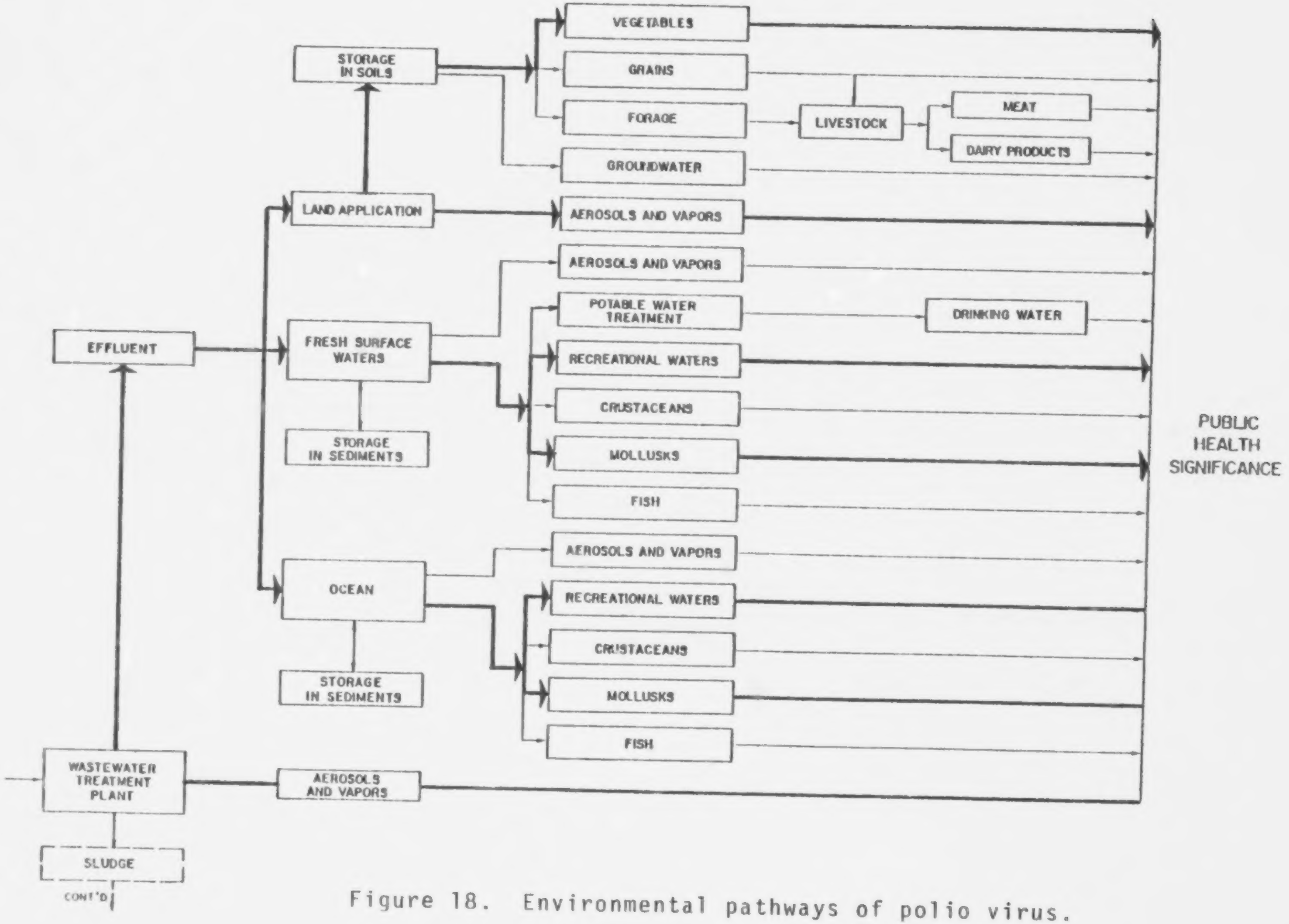

Figure 18. Environmental pathways of polio virus.

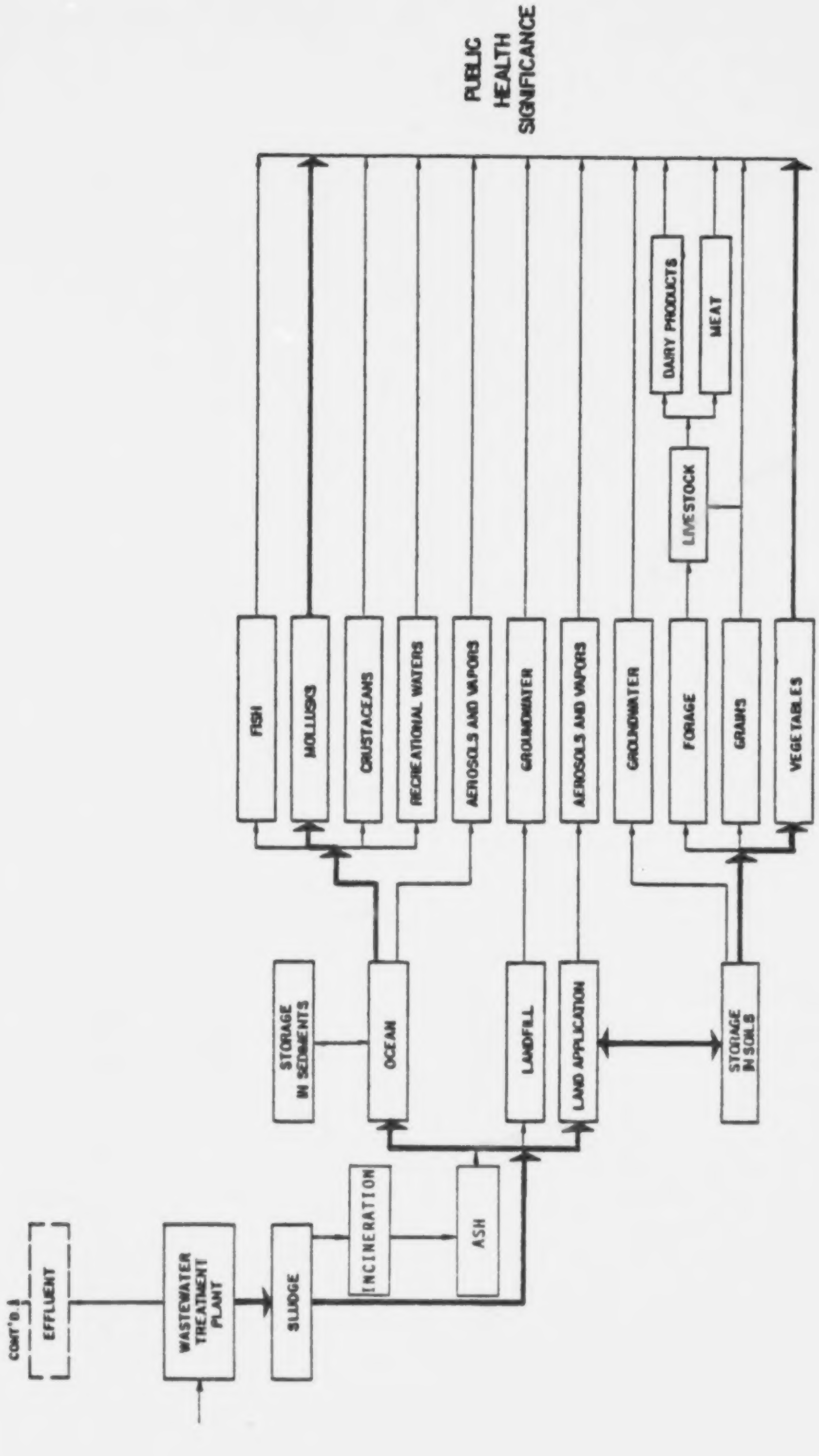

Figure 18. (continued)

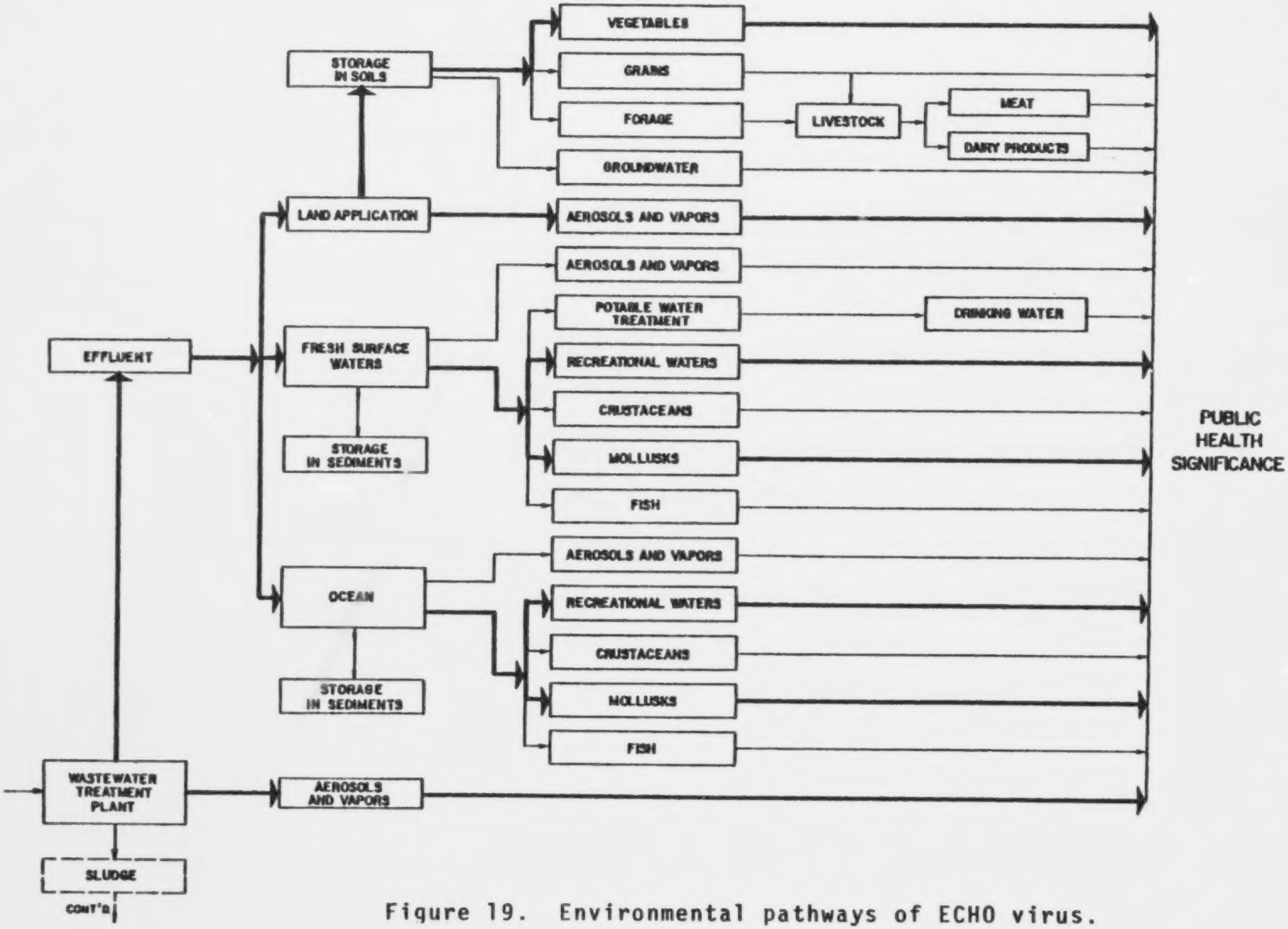

Figure 19. Environmental pathways of ECHO virus.

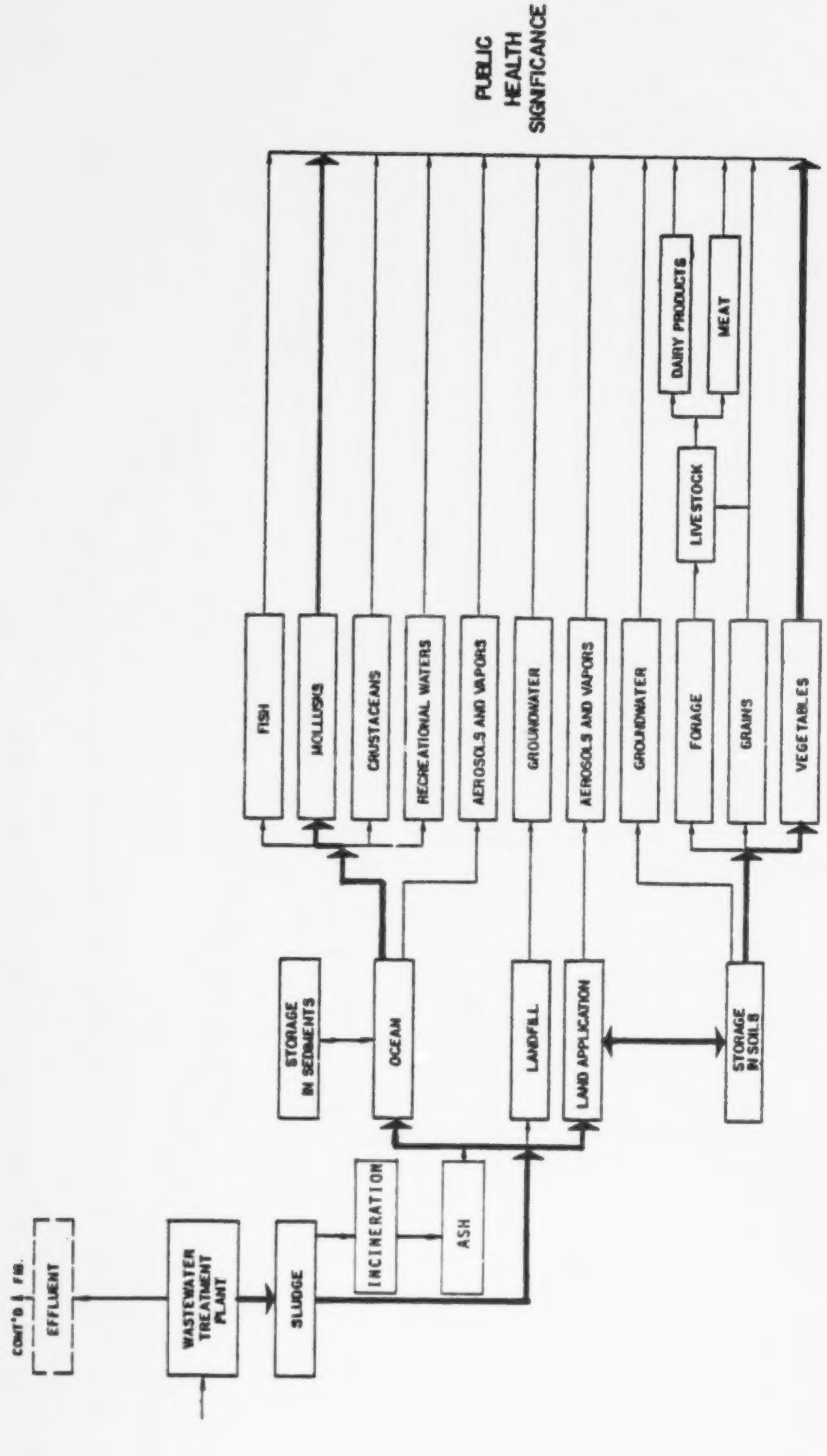

Figure 19. (continued)

With regard to viral contaminants, however, further research is recommended. A great deal of information is lacking concerning basic viral interactions in both the wastewater treatment plant and in the biosphere. Viruses are recognized as one of the most potentially hazardous and least understood components of municipal wastewaters. It is, therefore, advantageous to maximize virus removal in the wastewater treatment process. A significant amount of research is necessary before knowledge of viral contaminants is equal to that of other biological contaminants. Further research in the following wastewater and sludge treatment techniques is recommended with regard to viral behavior:

- Because of the widespread use of activated sludge processes, a study should be performed on the effects of various operating parameters on virus removal.

- It has been documented that activated carbon processes are capable of removing viruses. However, the exact mechanisms of removal and the factors that affect adsorption/desorption need further investigation so that viral removals can be maximized.

- The use of both pH extremes and chemical coagulation need to be further investigated due to the ability of these processes to remove and inactivate viruses. This information is necessary in order to identify and specify optimum treatment conditions.

- A substantial body of knowledge exists regarding the inactivation of viruses by chlorination; however, further investigation is needed into the mechanisms of inactivation, especially the methods by which adsorbed viruses may be exposed to the disinfectant. The types of solids that are capable of adsorbing and protecting virus from disinfection include: fecal matter, clays, hydrated metal oxides, carbonate and phosphate precipitates, activated carbon, activated sludge, and organic matter. The methods of viral destruction by chlorine that must be investigated include increased contact time and dosage. Some research has been conducted on the virus-inactivation capability of ozonation. However, due to the limited use of ozone for wastewater disinfection in this country, further research is given low priority. With the discovery of potential problems associated with the formation of chlorinated hydrocarbons by chlorination, study of virus inactivation mechanisms by ozone will be warranted.

Land application systems will be more widely used in the future for both the treatment and disposal of wastewaters and sludges; thus, the study of viral interactions in the soil matrix is a vital research area. Clearly there are many topics that must be investigated if a complete understanding of viral behavior in the environment is to be understood and regulated. The following subject areas are basic to understanding the implications of viral occurrence in wastewater treatment and sludges:

- The survival and fate of viruses in the soil systems require further research. Emphasis should be placed on the surface chemistry of viruses, how they are adsorbed by solids and soils, and mechanisms for desorption and inactivation.

- Further research is needed into the ecological factors that affect virus survival and activity in the aquatic environment. Especially recommended for study are shellfish interaction, concentrating effects, etc. In addition, it is critical that an appropriate indicator organism be located, if possible, to allow simple detection of the potential presence of viruses. Data from existing literature shows poor correlation between E. coli and virus die-off rates.

- Some work is currently being conducted on viruses in the aerosol pathway. However, this work needs to be consolidated to provide definitive information on the following:

    - The public health significance of airborne viruses and pathogenic bacteria from treatment plants as determined by epidemiologic and other health impact studies

    - Field methods for sampling, concentrating, and detecting airborne pathogens

- Survival of virus in the airborne state and the factors that influence survival
- Methods for controlling the generation of aerosols and transport of pathogens
- Development of an indicator organism for airborne virus and bacterial pathogens.

With the increasing reuse of wastewater in spray application systems, the aerosol pathway for virus will assume greater importance.

In many instances, much of the research necessary to completely understand viral behavior in the environment is dependent upon detection techniques that are not completely developed. In order to understand viral behavior in the wastewater treatment plant and in the ecosystem, certain basic subject areas should be researched:

- The cell culture system of virus analysis should be evaluated, and a reliable, straightforward analytical technique should be standardized. Presently, different laboratories using the same cell line give variable results for viral plating efficiencies. With standardization of techniques, different laboratories would be capable of assaying a test seed of virus and of obtaining essentially similar titers.
- Better methods for detecting, isolating, and assessing viruses are needed. Quantitative methods are required for detecting virus concentrations in sludges and soils.

SECTION II

REPORT TEXT

INTRODUCTION

FOREWORD

Since the public health disaster of Minimata, Japan, in the late 1950s and early 1960s (caused by ingestion of shellfish contaminated with methyl mercury) there has been a great surge in research concerning environmentally induced health effects on man. It is now generally accepted that the myriad of contaminants that is continuously discharged to the environment may produce both acute and chronic repercussions on public health through daily intake of air, water, and food. Recently, some forms of cancer, once thought to be of genetic etiology, have been projected to be caused or stimulated by environmental contaminants.

One area of critical consideration to public health is the treatment and disposal of municipal wastewater and sludges. Municipal wastewater systems have been the repositories of virtually every chemical contaminant known to or produced by man. The very nature of municipal wastewater streams makes them of critical importance when considering the environmentally induced health effects on man. Many questions must be clearly answered before a complete understanding of the situation may be gained.

What are the harmful constituents of municipal wastewater? How well does our present treatment technology remove these constituents? What eventually happens to them? What are their effects on the health and well-being of man? This project is an attempt to assemble and evaluate the existing literature pertinent to these questions. It is not an in-depth study. Rather, it is intended to serve as a basis for determining the critical areas of future research and remedial action.

OBJECTIVES

The major objective of this study was to conduct a comprehensive state-of-the-art literature search on the health effects of municipal wastewater treatment/disposal management systems. In satisfying the main objective, the following subordinate objectives were achieved with varying degrees of success:

- Selection of the public wastewater contaminants having health significance

- Determination of the ranges of concentrations of these contaminants contained in the influent to municipal wastewater treatment plants

- Assessment of the effectiveness of conventional wastewater and sludge treatment processes on the removal, modification, or inactivation of these contaminants

- Discussion of the potential hazards associated with effluent and treated sludge disposal practices in regard to the introduction of contaminants to the ecosystem

- Identification of the pathways by which contaminants associated with wastewater treatment/disposal operations can intersect with man

- Description of the chronic and acute effects of the contaminants within the human body

- Compilation of current research pertinent to each subject area, including project title, location, and chief investigator.

SCOPE

This project is an attempt to compile the published quantitative data available concerning the health effects associated with wastewater treatment and disposal. The assembled information includes data on the effectiveness of conventional treatment and disposal operations in reducing public health contaminant concentrations, as well as data on the transport of these contaminants through the environment back to man. The data have been organized in such a manner that the various pathways of pollutants to man can be evaluated for relative public health significance in order to establish necessary research priorities.

Wastewater treatment processes evaluated include conventional primary and secondary treatment processes; tertiary processes; and sludge dewatering, stabilizing, and destruction processes.

Wastewater disposal techniques evaluated include direct discharge to fresh or marine surface waters, land application, irrigation of crops, and recreational reuse in lakes.

A variety of pathways for potential contaminant intersection with man are investigated. These include contamination of food, air, and water by various routes through the biosphere.

The project was limited to a literature search and did not involve original research by the authors. Over 1500 literature references were reviewed and entered in the report bibliography. The information output of the project has been divided into two volumes as follows:

Volume I:	Executive summary Report text Report bibliography Current research
Volume II:	Annotated bibliography

The executive summary draws the main body of the report together. Mass balances through the environment were developed for some of the more toxic contaminants and expressed graphically, clearly demonstrating areas in which there is not enough information. The main body of the report consists of text and accompanying bibliography. This is followed by a section on current research that lists pertinent research in progress or slated to begin in near future.

APPROACH

This project was accomplished in four distinct phases. The first phase required was the location, review, and cataloging of pertinent literature by subject.

The second phase involved the evaluation of the literature acquired and the preparation of a draft report summarizing the current state of the art in each subject area. This phase was conducted from October 1975 to January 1976.

During the third phase, there was a critical review of the draft report by a panel of acknowledged experts who assembled for two days in late February 1976. This panel of experts included the following:

Name	Area of Expertise	Affiliation
Dr. Kenneth Y. Chen	Environmental Engineering	University of Southern California
Dr. Florence L. Harrison	Biology	University of California at Berkeley

Name	Area of Expertise	Affiliation
Dr. James J. Morgan	Water Chemistry	California Institute of Technology
Dr. Richard L. O'Brien	Pathology	University of Southern California
Dr. Daniel A. Okun	Environmental Engineering	University of North Carolina
Dr. Albert L. Page	Soil Science	University of California at Riverside
Dr. Lawrence W. Slanetz	Bacteriology	University of New Hampshire
Dr. Craig Wallis	Virology	Baylor College of Medicine

The fourth phase of the project entailed revising the draft to reflect the criticisms and recommendations of the foregoing experts. Also included in this final report rewrite was an extensive volume of literature that had been garnered after the draft report had been written. A brief summary of the individual tasks performed follows.

Library Research

Research assistants obtained copies of pertinent literature from several major university libraries, the Library of Congress, information retrieval systems (e.g., NTIS), and the SCS Engineers in-house library. Due to the breadth of the health effects topic, several constraints were placed upon the literature search in order to assure a workable, yet comprehensive, volume of information. Major constraints were as follows:

- Only literature published within the last ten years was reviewed except in unusual circumstances.
- Except in unusual circumstances, only literature directly addressing wastewater treatment and disposal and health effects was reviewed. There is, of course, a large volume of related literature (e.g., biological medical, meteorological) that could provide additional insights; however, this peripheral literature was excluded from the investigation.

CORRESPONDENCE

To obtain the very latest scientific information available as well as to find out in which areas research is presently being conducted, letters were sent soliciting literature and data to the following information sources:

- State regulatory and public health agencies,
- Major sanitary districts known to be conducting research, and
- Universities conducting related research.

The responses from these sources have been incorporated into all phases of the project.

Data Organization

A large matrix was prepared listing contaminants vertically and treatment processes, disposal alternatives, and pathways to man horizontally. All pertinent references were indexed in one or more of the matrix squares. The research team could thus quickly obtain the references required to summarize a particular subject area.

Report Organization

This report is organized by pathway rather than by contaminant, so that various public health impairing contaminants can be traced through wastewater treatment plants, through the biosphere, and finally back to man. Figure 20 illustrates the alternate pathways that various contaminants may follow to reach man after leaving the wastewater treatment plant.

Each major wastewater treatment step and disposal method is treated in a separate subsection of the report. The reader is able, for instance, to go to the Activated Sludge Treatment section to review the effect of this process upon all the contaminants. The report does not, however, contain a separate section for each contaminant. If the reader is interested in mercury, for example, he must skip through the report and read about the fate of mercury during individual treatment processes, when released to the biosphere, and the consequent health effects on man. To facilitate the task of tracing a single contaminant through the report, each section is divided into subsections by contaminant type.

The selection of wastewater contaminants for consideration in this study was difficult because of the many public health impairing constituents found in wastewater. Several of the

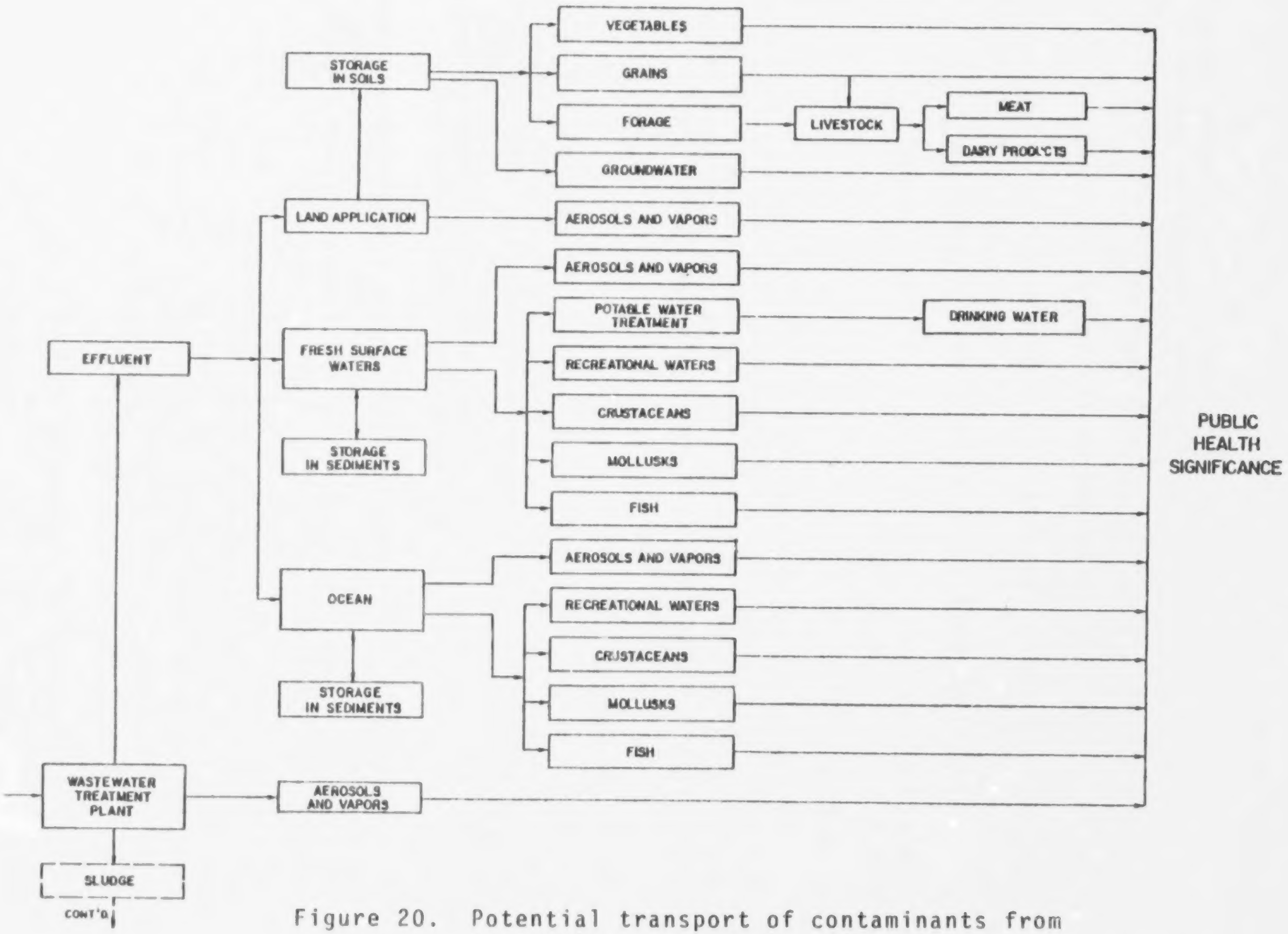

Figure 20. Potential transport of contaminants from wastewater management systems to man.

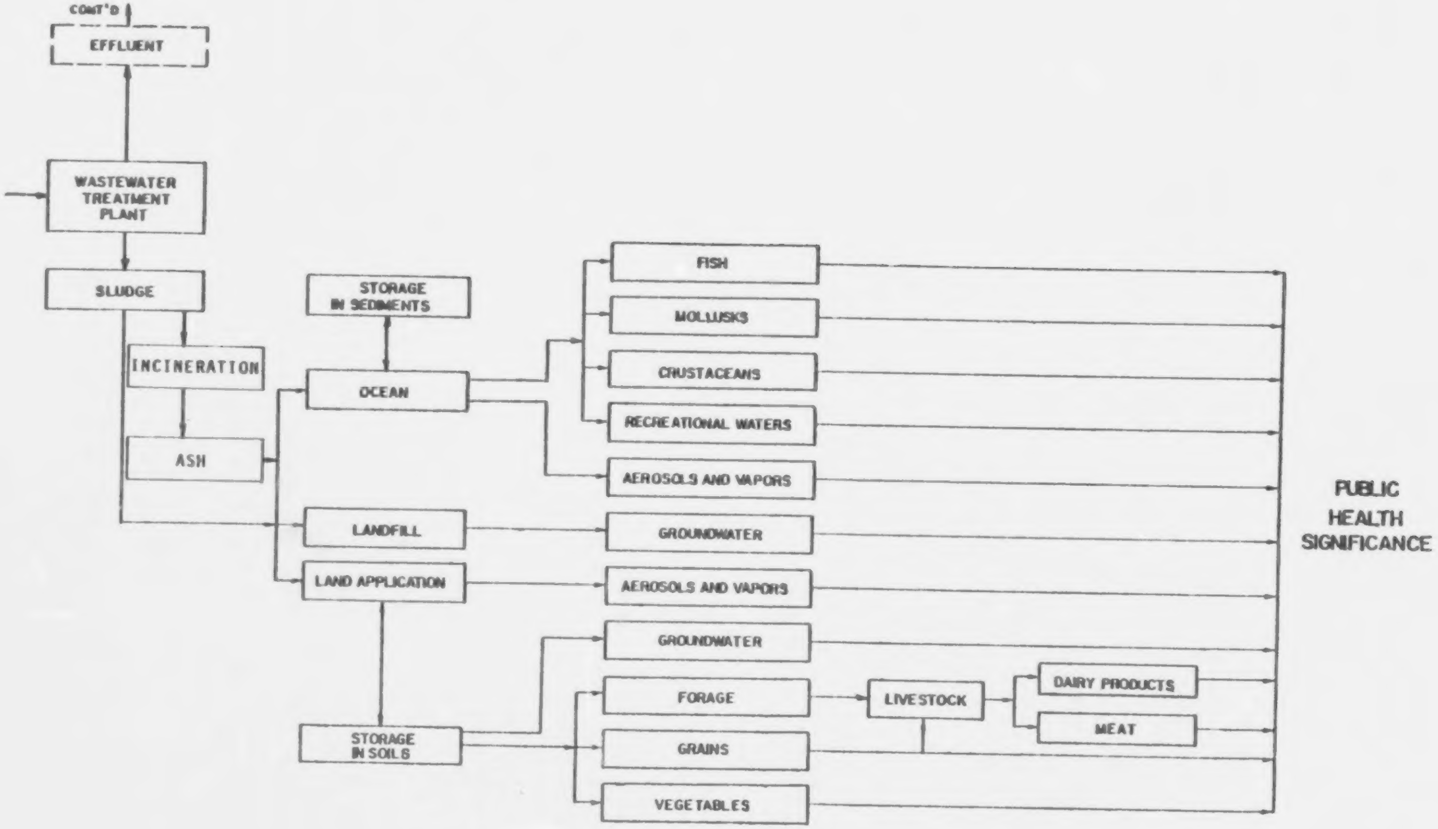

Figure 20. (continued)

traditional wastewater parameters (BOD, suspended solids, etc.) pose no direct threat to public health, although some of the direct health-impairing contaminants may be associated with these traditional wastewater parameters. Therefore, some of the traditional parameters are included.

Within each section of this report - tracing wastewater contaminants through municipal wastewater treatments, through the environment, and finally detailing the epidemiological and pathological effects on man - information is organized and presented in the following contaminant groups:

1. Water Quality Parameters
2. Elemental Contaminants
3. Biocidal Contaminants
4. Synthetic/Organic Contaminants
5. Biological Contaminants

The water quality parameters group contains those water measurements that are traditionally associated with wastewater treatment systems and water quality, as well as those contaminants that do not readily fit in any other category. Table 1 lists these water quality parameters.

TABLE 1 . WATER QUALITY PARAMETERS

1. Ammonia
2. Biochemical Oxygen Demand (BOD)
3. Chemical Oxygen Demand (COD)
4. Nitrates
5. Nitrites
6. Phosphates
7. Suspended Solids
8. Total Dissolved Solids (TDS)
9. Total Organic Carbon (TOC)

Most of the parameters listed in Table 1 do not pose a direct threat to public health but may be related to public health impairing contaminants. For example, heavy metals or viruses may be adsorbed to the surface of suspended solids and may be transported through the wastewater treatment plant and the biosphere in this manner. However, the nitrogenous compounds (ammonia, nitrates, and nitrites) may directly threaten public health.

The elemental contaminants group, presented in Table 2, primarily consists of heavy metals.

TABLE 2. ELEMENTAL CONTAMINANTS

1. Aluminum
2. Antimony
3. Arsenic
4. Barium
5. Beryllium
6. Boron
7. Cadmium
8. Chromium
9. Cobalt
10. Copper
11. Germanium
12. Iron
13. Lead
14. Manganese
15. Mercury
16. Molybdenum
17. Nickel
18. Selenium
19. Thorium
20. Tin
21. Uranium
22. Vanadium
23. Zinc

Many of these elemental contaminants are required in trace quantities by the human body for normal metabolic functions, yet higher levels of these trace elements may cause significant health problems.

Biocidal contaminants are those contaminants normally used to control insect or desease vectors. Table 3 lists these biocidal contaminants.

TABLE 3. BIOCIDAL CONTAMINANTS

1. DDT
2. DDD
3. DDE
4. Aldrin
5. Dieldrin
6. Endrin
7. Chlorinated hydrocarbons
8. Arsenated hydrocarbons
9. Organonitrogen pesticides
10. Organophosphorus pesticides
11. Herbicides
12. Soil sterilants

DDT, DDD, DDE, aldrin, dieldrin, and endrin are all chlorinated hydrocarbons; they were considered separate from the chlorinated hydrocarbon classification because the literature often dealt with these specific pesticides as individual entities.

The synthetic/organic contaminants group includes many synthetically produced organic chemicals that have found their way into water systems. For the purposes of this report, information was gathered on any synthetic/organic chemicals in wastewater treatment processes or the biosphere. A selection of such contaminants recently identified by the Environmental Protection Agency in an evaluation of organic compounds in the New Orleans area water supply is listed in Table 4.

Traditionally, biological contaminants have received the most attention in wastewater treatment, since these contaminants may directly cause public infection. Those biological contaminates considered in this study are listed in Table 5.

TABLE 4. ORGANIC COMPOUND IDENTIFICATION NEW ORLEANS AREA WATER SUPPLY STUDY (1323)

1. Acetaldehyde
2. Acetone
3. Alkylbenzene-C_2 isomer
4. Alkylbenzene-C_2 isomer
5. Alkylbenzene-C_2 isomer
6. Alkylbenzene-C_3 isomer
7. Alkylbenzene-C_3 isomer
8. Alkylbenzene-C_3 isomer
9. Atrazine (2-chloro-4-ethylamino-6-isopropylamino-*s*-triazine)
10. Deethylatrazine (2-chloro-4-amino-6-isopropylamino-*s*-triazine)
11. Benzyl butyl phthalate
12. Bromodichloroethane
13. Bromoform
14. Butanone
15. Carbon disulfide
16. Carbon tetrachloride
17. bis-2-Chloroethyl ether
18. Chloroform
19. bis-2-Chloroisopropyl ether
20. n-Decane
21. Decane-branched isomer
22. Dibromodichloroethane isomer
23. Dibromochloromethane
24. Dibutyl phthalate
25. 2,6-Di-t-butyl-*p*-benzoquinone
26. Dichlorobenzene isomer
27. 1,2-Dichloroethane
28. Dichloromethane
29. Dieldrin
30. Diethyl phthalate
31. Di(2-ethylhexyl) phthalate
32. Dihexyl phthalate
33. Dihydrocarvone
34. Diisobutyl phthalate
35. Dimethyl phthalate
36. Dioctyl adipate
37. Dipropyl phthalate
38. n-Dodecane
39. Endrin
40. Ethanol
41. *o*-Ethyltoluene
42. *p*-Ethyltoluene

TABLE 4 (continued)

43. 1, 2, 3, 4, 5, 7, 7-Heptachloronorbornene

44. Heptachloronorbornene isomer

45. Hexachloro-1,3-butadiene

46. Hexachloroethane

47. Isophorone

48. Limonene

49. Methanol

50. Methylbenzoate

51. 3-Methylbutanal

52. 2-Methylpropanal

53. n-Nonane

54. n-Pentadecane

55. Tetrachloroethane isomer

56. Tetrachloroethylene

57. n-Tetradecane

58. Toluene

59. 1,1,2-Trichloroethane

60. 1,1,2-Trichloroethylene

61. n-Tridecane

62. Trimethyl-trioxo-hexahydrotriazine isomer

63. Triphenyl phosphate

64. n-Undecane

65. Undecane-branched isomer

TABLE 5. BIOLOGICAL CONTAMINANTS

1. Adenovirus
2. Coxsackie virus (A&B)
3. ECHO virus
4. Hepatitis virus
5. Polio virus
6. Coliforms
7. Salmonella
8. Shigella
9. Leptospira
10. *Escherichia coli*
11. *Francisella tularensis*
12. *Vibrio cholerae*
13. Mycobacterium
14. Streptococci
15. Staphylococcus aureus
16. *Clostridium botulinum*
17. *Clostridium perfringens*
18. *Listeria monocytogens*
19. *Erysipelothrix rhusiopathia*

20 Yeasts

21. Protozoa
22. Parasitic worms

WASTEWATER TREATMENT SYSTEMS

WASTEWATER INPUTS

INTRODUCTION

Untreated wastewater input composition is the first point of interest in determining the pathways pollutants may follow from wastewater management systems back to man. In addition to domestic sewage, input sources may include various industrial wastes, storm water, and groundwater infiltration, in various combinations. Moreover, the input proportions of an individual system change with time; pollutant concentrations and volumes vary hourly, daily, weekly, and seasonally.

Research surveyed regarding input concentrations to municipal treatment facilities is presented in Table 6 . Substantial literature concerning input compositions is available for water quality parameters and elemental and biological contaminants. With few exceptions, however, input concentrations of biocidal and synthetic-organic contaminants have not been investigated. It was therefore difficult to determine the effect of subsequent treatment processes upon such contaminants. Regulations for industrial discharges of complex mixtures of organic compounds generally only require reporting of the BOD or COD, suspended solids, and similar water quality parameters. In many instances, industrial concerns themselves may not know the detailed composition of their waste streams.

Since 1972, the development of the National Pollution Discharge Elimination System (NPDES) permit program has changed the typical input composition to municipal sanitary systems through increased restrictions on inputs from industrial waste. In particular, these restrictions have reduced the amount of heavy metal contaminants entering municipal treatment systems.

Municipal treatment systems that process only domestic wastes and that have no excessive infiltration show predictable diurnal and seasonal patterns. However, most urban sanitary systems and many rural systems also contain industrial wastes. Composition and flow volumes may vary significantly as a function of these industrial inputs. Food processing plants can contribute seasonally high BOD discharges; large metal platers and metal finishers may periodically contribute high metal waste concentrations. Certain situations require particular attention when evaluating potential adverse effects on human health, including:

TABLE 6. LITERATURE REVIEWED PERTAINING TO THE COMPOSITION OF WASTEWATER INPUTS TO MUNICIPAL TREATMENT SYSTEMS

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	47, 165, 517, 745, 801, 829, 907, 1052, 1084, 1254, 1389, 1402, 1510
BOD	39, 124, 321, 483, 517, 602, 690, 745, 801, 907, 1052, 1104, 1215, 1389, 1402, 1403, 1510
COD	39, 124, 249, 321, 483, 616, 801, 802, 907, 1084, 1104, 1254, 1389, 1402, 1510
Chlorides	39, 47, 301, 907, 1084
Cyanides	801
Fluorides	801
Nitrates	47, 165, 517, 745, 829, 907, 1052, 1071, 1084, 1254, 1389, 1402
Nitrites	165, 517, 745, 801, 829, 907, 1052, 1084, 1254, 1389, 1402, 1510
Oil & grease	1153
Phosphates	39, 47, 124, 245, 249, 301, 517, 606, 745, 801, 829, 907, 1084, 1254, 1389, 1402, 1510
Suspended solids	47, 249, 321, 616, 745, 1004, 1084, 1104, 1254, 1389, 1402, 1403, 1510
Total dissolved solids	2, 801, 1389
Total organic carbon	124, 249, 616, 1084, 1389
Other (general)	10, 616, 745, 1051
Elemental Contaminants	
Aluminum	63, 249, 801, 1140
Arsenic	39, 801

TABLE 6 (continued)

Contaminant	Reference Number
Barium	39, 801
Cadmium	39, 78, 293, 801, 802, 982
Chromium	39, 249, 293, 801, 802, 969, 982, 984
Cobalt	249, 517, 801, 982, 1516
Copper	39, 205, 249, 293, 517, 801, 982, 1516
Iron	63, 249, 517, 907, 982, 1140, 1516
Lead	39, 205, 249, 801, 982, 1018
Manganese	249, 517, 801, 907, 982, 1516
Mercury	39, 803, 982
Molybdenum	801, 1516
Nickel	205, 293, 517, 801, 982
Selenium	39, 801, 802
Tin	801
Zinc	39, 63, 205, 249, 293, 801, 1516
Synthetic/Organic Contaminants	621, 881
Biological Contaminants	
Coliforms	249, 307, 483, 745, 801, 1104, 1182
ECHO virus	1182
Fecal streptococci	307, 458, 801, 1182
Mycobacterium	428
Parasitic worms	428
Polio virus	1182

TABLE 6 (continued)

Contaminant	Reference Number
Protozoa	428
Salmonella	428, 672
Shigella	428
Virus	380, 428, 1104

- Large hospital complexes connected to small sanitary systems

- Situations in which large amounts of waste from metal platers and metal finishers discharge to the system

- Tanneries discharging to the system

- Petrochemical and related complexes

- Biocide manufacturing facilities

- Speciality chemical manufacturing or formulating facilities

- Combined or heavily infiltrated systems that bypass excess flow without adequate treatment.

WATER QUALITY PARAMETERS

The water quality parameters that characterize wastewater entering into a sanitary treatment plant have been extensively studied. Concentration ranges recorded in the reviewed literature are presented in Table 7 . Atypical situations that lie outside these ranges are known to occur under special conditions. Such exceptions can usually be attributed to excessive industrial inputs, excessive infiltration, or unusual characteristics of the fresh-water supply to the service area.

Diurnal flow patterns reported by Nomura and Young (1004) show that suspended solids concentrations in raw sewage are directly related to sewage flow rates. The suspended solids content of the raw sewage studied ranged from 20 mg/ℓ in the late night flow (attributed to infiltration) up to a high of 350 mg/ℓ during the peak daily sewage flow rate.

Storm and municipal systems show considerable variation in flow rate as a function of time; consequently, contaminant concentrations vary greatly (10, 745, 1051). Water quality parameter variations due to storm system input will be a function of:

- Normal concentrations during dry weather flow
- Amount and duration of precipitation
- Time since the last rain
- Season

ELEMENTAL CONTAMINANTS

The type and amount of elemental contaminants contained in any treatment system input will depend primarily upon the type and amount of industrial wastes entering that system. A great

TABLE 7. SUMMARY OF WATER QUALITY PARAMETERS CHARACTERIZING WASTEWATER INPUT TO MUNICIPAL TREATMENT FACILITIES*

Constituent	Range, mg/ℓ
Ammonia nitrogen	8 to 50
Total nitrogen	20 to 85
Organic-nitrogen	5 to 32
Nitrate-nitrogen	0 to 3
Nitrite-nitrogen	0 to 1
Chloride	25 to 203
Oil and grease	1 to 50
Total phosphorus	4 to 50
Phosphate as PO_4	5 to 50
Inorganic phosphorus	8 to 13
Organic phosphorus	1 to 5
BOD	30 to 600
COD	100 to 1000
Suspended solids	30 to 350
Total dissolved solids	250 to 1400

*As reported in the literature reviewed.

deal of material is available on industrial discharges to municipal systems. A summary range of elemental influent characteristics as reported in the literature is presented in Table 8 . The Interim Drinking Water Standards have been included to provide a standard of comparison.

The literature reviewed did not provide a comprehensive survey of the sources of these metallic contaminants, but the following factors appear to be important:

- Input water composition
- Input water scaling and corrosion potential
- Type and age of domestic water piping systems
- Type and amount of industrial discharges
- Type of municipal system (combined or separate storm water).

The relative significance of these factors will again depend upon the specific site. The results of a study by Davis and Jacknow (293) of the 12 municipal treatment plants in New York City (presented in Table 9) give some idea of the ratio of contributions from various sources that might be expected in a large urban area. This same study contains source breakdowns for individual plants. Input from the residential sector contained concentrations exceeding the Interim Primary Drinking Water Standards by a factor of 2.

A knowledge of the physical and chemical forms of elemental contaminants is necessary in order to adequately evaluate potential public health effects. In particular, it is necessary to know whether these elements are present as particulate materials or soluble species. Data of this type are apparently unavailable at this time.

BIOLOGICAL CONTAMINANTS

Information available in the literature concerning input concentrations of biological contaminants generally addresses primary indicator organisms rather than specific pathogens. A summary range of biological contaminants reported in this literature is presented in Table 10.

TABLE 8. SUMMARY OF THE CONCENTRATION RANGES OF ELEMENTAL CONTAMINANTS FOUND IN WASTEWATER INPUTS TO MUNICIPAL TREATMENT SYSTEMS*

Constituent	Range mg/ℓ	Interim Drinking Water Standards
Aluminum	0.3 to 3.0	--
Antimony	No data in literature	--
Arsenic	0 to 0.02	.05
Barium	0 to 0.02	1
Beryllium	No data in literature	--
Boron	0.5 to 3	--
Cadmium	0.01 to 0.2	.01
Chromium	0.01 to 0.3	.05
Cobalt	No data in literature	--
Copper	0.01 to 0.5	--
Germanium	No data in literature	--
Iron	0.5 to 6.5	--
Lead	0 to 1	.05
Manganese	0.05 to 0.15	--
Mercury	0.0002 to 0.003	.002
Molybdenum	No data in literature	--
Nickel	0.05 to 0.5	--
Selenium	0 to 0.11	.01
Thorium	No data in literature	--
Tin	No data in literature	--
Uranium	No data in literature	--
Zinc	0.01 to 2.10	--

*As reported in the literature reviewed.

TABLE 9. AVERAGE HEAVY METAL LOADINGS AND PROBABLE SOURCES FOR TWELVE NEW YORK CITY MUNICIPAL TREATMENT PLANTS (293)

Metals	Total Loading (lb/d)	Electroplating and Photoengraving (lb/d)	Percentage of total	Estimate of Residential Contribution (lb/d)	Percentage of Total	Other Industries, Urban Storm water, and Other Contributions (lb/d)	Percentage of total
Copper	3,820	611	15	1,440	38	1,770	46
Chromium	2,340	345	14	640	27	1,350	58
Nickel	1,870	1,010	53	640	34	220	12
Zinc	10,440	830	7	1,680	16	7,930	76
Cadmium	341	65	19	128	38	148	43

TABLE 10. SUMMARY OF THE CONCENTRATION RANGES OF BIOLOGICAL CONTAMINANTS FOUND IN WASTEWATER INPUTS TO MUNICIPAL TREATMENT SYSTEMS*

Contaminant	Range/100 mℓ
Total coliforms	1×10^6 to 4.6×10^7
Fecal coliforms	3.4×10^5 to 4.9×10^7
Fecal streptococci	6.4×10^4 to 4.5×10^6
Virus	5 to 100,000 virus units/ℓ

*As reported in the literature reviewed.

Pound and Crites (1104) reported that raw municipal sewage contains from 10^6 to 10^8 total coliforms and from 480 to 1,677 pfu/ℓ of enteric viruses. The average enteric virus density in domestic sewage reported by the ASCE (380) was approximately 500 virus units/100 mℓ. Coliform densities averaged 4.6×10^7/100 mℓ. Analyses of wastewater reported by Kampelmacher and Jansen (672) show that salmonella is regularly present.

Viral concentrations in raw wastewater reported by Foster and Engelbrecht (428) range from 5 to 100,000 viral pfu/ℓ, with reported seasonal variations of a mean of 4,000 pfu/ℓ during warm months and a mean of 200 units/ℓ during cold months.

Fecal streptococci contaminants present in the domestic wastewater of seven communities were studied by Geldreich and Kenner (458). Figures obtained from this study are presented in Table 11.

TABLE 11. DISTRIBUTION OF FECAL STREPTOCOCCI IN DOMESTIC WASTEWATER AND STORM-WATER RUNOFFS (458)

Water Source	Densities/100 mℓ				Occurrence (%)			
	Fecal Coliforms	Fecal Streptococci	Ratio FC/FS	Total Strains Examined	Entero-cocci	*S. bovis* *S. equinus*	Atypical *S. faecalis*	*S. faecalis liquifaciens*
Domestic wastewater								
Preston, Ida.	340,000	64,000	5.3	39	79.5	None	None	20.5
Fargo, N. D.	1,300,000	290,000	4.5	50	100.0	None	None	None
Moorehead, Minn.	1,600,000	330,000	4.9	50	90.0	10.0	None	None
Cincinnati, Ohio	10,900,000	2,470,000	4.4	428	71.5	2.8	1.6	24.1
Lawrence, Mass.	17,900,000	4,500,000	4.0	50	84.0	4.0	None	12.0
Monroe, Mich.	19,200,000	700,000	27.9	70	78.6	1.4	4.3	15.7
Denver, Colo.	49,000,000	2,900,000	16.9	70	85.7	11.4	2.9	None
Storm water								
Business district	13,000	51,000	0.26	1,476	78.5	1.6	1.2	18.8
Residential	6,500	150,000	0.04	1,158	80.0	0.5	1.4	18.1
Rural	2,700	58,000	0.05	445	87.4	0.5	0.2	11.9

PRIMARY TREATMENT

INTRODUCTION

Primary treatment is intended to physically remove settleable solids and most of the discrete suspended and floating solids from the municipal wastewater stream preparatory to secondary treatment. In addition, primary treatment removes a limited portion of the soluble constituents.

In primary treatment, the wastewater influent is divided into three output pathways: primary effluent, primary sludge (including grit, screenings, and precipitated matter), and aerosols. Effluent from primary treatment can be directly discharged (until 1977), discharged after disinfection (again until 1977), or treated by a secondary process. Primary sludge is normally subject to additional processing. At some ocean coastal sites, however, the sludge is discharged without further treatment. Aerosols are rarely a problem, due to the absence of excessive turbulence.

The information reviewed during this study is tabulated in Table 12. Most research conducted to date concerns the removal by primary treatment of water quality parameter constituents. It is only in recent years that researchers have examined the effect of primary treatment on various public health impairing contaminants.

WATER QUALITY PARAMETERS

For several decades, extensive work has been reported concerning general contaminant removal efficiencies for primary treatment. Removal varies widely, depending upon the physical and chemical characteristics of the wastewater, the proportion of settleable solids, concentrations of the solids, and detention time. Mitchell (922) reported the general performance efficiencies that could be expected for typical primary treatment (Table 13). Removals achieved during a three-year study of an operational primary treatment system are shown in Table 13.

TABLE 12. LITERATURE REVIEWED PERTAINING TO PRIMARY WASTEWATER TREATMENT

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	922
BOD	393, 622, 745, 922, 1404, 1435, 1547
COD	483, 622, 896, 922, 1435
Cyanides	922
Oil and grease	483, 810, 922
Phosphates	896
Suspended solids	393, 483, 622, 745, 896, 922, 1435, 1547
Total organic carbon	483
Other (general)	531, 622
Elemental Contaminants	
Arsenic	922
Cadmium	228, 443, 922, 1063
Chromium	228, 896, 922
Copper	228, 443, 896, 922, 1063
Iron	228, 896
Lead	228, 922, 1063
Manganese	228
Mercury	228, 922
Nickel	228, 922, 1063
Selenium	1114
Zinc	228, 896, 922, 1063

TABLE 12 (continued)

Contaminant	Reference Number
Biological Contaminants	
Bacteria	79, 161, 1256
Coliforms	161, 483, 568, 717, 896, 1256, 1282
Coxsackie virus (A & B)	161
Escherichia coli	568, 896
Streptococci	568
Hepatitis virus	1261
Mycobacterium	161, 428
Parasitic worms	161, 420, 428
Polio virus	97, 161
Protozoa	161, 428
Salmonella	161, 420, 428, 568, 1261
Shigella	161, 1261
Vibrio cholerae	161
Virus	79, 92, 95, 96, 97, 100, 101, 161, 382, 428, 468, 1009, 1256
Other (general)	161, 1261

TABLE 13. PRIMARY SETTLING TANK PERFORMANCE (922)

Parameter	Primary Influent (mg/ℓ)	Primary Effluent (mg/ℓ)	Percent Removal
COD	539	315	42
BOD_5	269	165	39
Suspended Solids	279	103	64
Oil and Grease	72	28	61
Ammonia Nitrogen	34	20	41

When used as a coagulant during primary treatment, lime is most effective in reducing certain water quality parameter constituents. When lime addition of 350 mg/ℓ was followed by air flotation, Mennell et al. (896) reported the following removals: turbidity, 98.5 percent; suspended solids, 95 percent; COD, 60 percent; and total phosphorus, 99 percent. Total nitrogen removal varied between 10 and 20 percent. Lower lime dosages provided proportionally lower removal percentages. The practice of dosing primary clarifiers with chemical coagulants will probably increase, as municipalities attempt to cost effectively meet federal and state water quality standards.

ELEMENTAL CONTAMINANTS

Recent interest in elemental contaminants, particularly trace metals, has prompted investigation into the partitioning of these contaminants in the wastewater treatment stream. Primary treatment receives elemental contaminants in a variety of forms, e.g., soluble, insoluble, and complexed. The concentrations of each form vary intrinsically as a complex function of such factors as influent metal concentration, pH, and ligand concentration.

Mitchell (922) reported removal efficiencies for primary settling over a three-year period at the Hyperion treatment plant, Los Angeles, California, as shown in Table 14.

TABLE 14. PRIMARY TREATMENT REMOVAL OF METAL ELEMENTS (922)

	Primary Influent(mg/ℓ)	Primary Effluent(mg/ℓ)	Percent Removal
Copper	0.39	0.25	36
Zinc	0.66	0.42	36
Nickel	0.30	0.24	20
Lead	0.03	0.07	--
Arsenic	0.015	0.017	--
Cadmium	0.01	0.02	--
Chromium	0.55	0.37	32

No explanation was offered to account for the anomaly of increased lead, arsenic, and cadmium concentrations.

Chen and Lockwood (228), also working with the Hyperion treatment plant, discussed the partitioning of trace metals with particulates as a function of particle size. It was reported that in primary effluent, more than 70 percent of the total trace metals content is associated with particulates as opposed to the soluble ionic form. However, only 10 to 20 percent of the Ni, Pb, or Mn was associated with the particulate fraction. Similar concentrations of metals were found on the larger (44 μm) and the smaller (0.2 μm) particles. However, particles as large as 44 μm will settle about 200 times as rapidly as 3-μm particles. More efficient removals of elemental contaminants could be expected by the use of either increased wastewater detention time, or chemical precipitants (e.g., lime) for particle coagulation. Trace metal removals reported (896) at a lime dose of 388 mg/ℓ approached 100 percent for chromium and copper, 97 percent for iron, and 94 percent for zinc. Molybdenum was not effectively removed by this treatment.

BIOLOGICAL CONTAMINANTS

Primary sedimentation usually removes less than 50 percent of coliform and pathogenic bacteria from sewage and is relatively ineffective in removing viruses and protozoa. Literature concerning the removal of water-borne pathogens by primary treatment processes reports a varying degree of efficiency, depending in part on the type of pathogen studied. Table 15 (161, 428) highlights the results of Bryan's investigation of pathogen survival during primary treatment.

In their literature review, Foster and Englebrecht (428) reported isolation of salmonella from six of seven different primary effluent samples. The raw sludge also contained members of this genus, with 19 of 20 samples tested as positive for salmonella organisms. Tubercle bacilli were reduced about 50 percent in the wastewater stream during sedimentation. It was concluded that bacterial pathogens are ineffectively removed from wastewater by primary settling; furthermore, the process produced a sludge that, without further treatment, constitutes a health hazard.

Amoebic cysts and parasitic worm ova are also ineffectively removed by primary treatment, due to their low specific densities and resulting buoyancy. Ascaris ova are an exception: Foster and Engelbrecht (428) reported 100 percent settlement of these ova into the sludge within 15 min.

Viruses, because of their size (.02 to .3 μm), are rarely removed by sedimentation, except for those that are associated with wastewater solids. The nature of the surface chemistry of

TABLE 15. SURVIVAL OF PATHOGENS DURING PRIMARY TREATMENT (161, 428)

Pathogen	% Removal
Salmonella typhi	>50
Salmonella spp.	0-15
Streptococcus faecalis	<50
Mycobacterium	48-57
Enteroviruses	no reduction
Polio viruses	no reduction
Coxsackie viruses	<50
Endamoeba histolytica	0-50
Ascaris ova	0-100
Trichuris trichiura	>50
Taenia saginata	0-50

viral units suggests that adsorption depends strongly on the chemical environment, would be expected to vary according to changes in input water chemistry, and would possibly be affected by chemical additions.

The removal of viruses by primary settling has been both researched and reviewed extensively by Berg (92, 95, 97, 100). Berg described a project (95) in which only one to two-thirds of the viral particles had settled out in one day, although 75 percent of the suspended solids had settled. In this review, Berg discussed several additional studies on viral removal by primary treatment; all failed to mention detention time, and more importantly, the levels of virus in the incoming sewage were not related to those in the effluent.

Although primary settling alone will not effectively reduce the pathogen content of wastewater, dosing primary settling tanks with chemical coagulants does show some promise in this regard. Chemical precipitation, when used during primary treatment, is capable of removing as much as 99.99 percent of the virus suspended in water, effected through the formation of a coagulant-cation-virus complex. Elevated pH levels attained during lime treatment also result in substantial reductions in viral numbers (468). Lime coagulation during primary treatment brings remarkable reductions of coliform density as well. A 99.9 percent coliform reduction was measured at a lime dose of 450 mg/ℓ (896).

SECONDARY TREATMENT: ACTIVATED SLUDGE

INTRODUCTION

The activated sludge process involves the growth of microorganisms in a reactor. This effects partial biological degradation of organic compounds in wastewater to simpler organic compounds, carbon dioxide, water, microorganisms, and energy (393). The basic process requires two equipment components: aeration tanks and clarifiers. Active biological sludge is separated from the effluent in a clarifier and recycled to an aeration tank.

Activated sludge, the most popular wastewater secondary treatment process, has been extensively studied, as indicated by Table 16 . Most research has focused on water quality parameters such as BOD, COD, and suspended solids. The data are usually presented as percent removal, with removal efficiency determined by difference in influent and effluent concentrations. Removal of a specific contaminant can be accomplished by separation into the sludge or by degradation through biological activity. Aerosol generation from the aeration tank is also a possible contaminant pathway.

In view of possible health effects, the difference between separation and degradation can be significant. If the treatment process merely partitions a particular contaminant into the sludge or air, it remains available for migration back to man. In contrast, biological degradation can terminate the contaminant pathway or transform the potentially harmful substance into a nontoxic form. The separation and degradation components of the removal process are often not distinguished in the activated sludge literature.

WATER QUALITY PARAMETERS

Past research on activated sludge processes has concentrated on water quality parameters, with primary emphasis on BOD, COD, and suspended solids. Because of the tremendous volume of literature associated with BOD and suspended solids as indicators of activity, and the absence of direct health effects from these pollutants, this report placed greater emphasis on literature dealing with chemical and biological contaminants of more direct public health concern.

TABLE 16. LITERATURE REVIEWED PERTAINING TO ACTIVATED SLUDGE

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	4, 10, 59, 64, 113, 196, 277, 351, 352, 364, 385, 454, 614, 618, 799, 922, 1115, 1333, 1351
BOD	3, 10, 113, 146, 196, 209, 276, 352, 364, 560, 614, 623, 745, 799, 814, 864, 875, 922, 1003, 1084, 1240, 1337, 1360, 1435, 1493
COD	196, 277, 364, 560, 601, 623, 668, 799, 922, 1084, 1158, 1333, 1493
Chlorides	114, 819, 899, 1062
Cyanides	623, 824, 922, 1062
Fluorides	1062
Nitrates	10, 59, 64, 113, 122, 352, 364, 375, 447, 454, 1351
Nitrites	10, 122, 352, 454, 1351, 1516
Oil and grease	810, 922, 1062, 1516
Phosphates	10, 13, 62, 63, 64, 113, 114, 196, 277, 352, 364, 375, 410, 454, 510, 513, 555, 601, 614, 763, 785, 814, 899, 922, 1213, 1230, 1337, 1341, 1349
Suspended solids	3, 10, 113, 146, 196, 227, 352, 560, 623, 668, 745, 922, 1084, 1435, 1516
Total dissolved solids	196, 276, 352, 500, 899
Total organic carbon	10, 113, 114, 352, 615, 1158, 1333, 1516
Other (general)	385, 391, 393, 531, 602, 622, 1310, 1333
Biological Contaminants	
Aluminum	513, 814, 1004

TABLE 16 (continued)

Contaminant	Reference Number
Arsenic	1062
Barium	1062
Boron	352, 899, 1062
Cadmium	227, 228, 230, 922, 982, 992, 1004, 1062, 1063, 1323
Chromium	69, 71, 227, 228, 623, 922, 940, 969, 982, 1062, 1323
Cobalt	353, 982
Copper	69, 71, 227, 228, 230, 623, 875, 922, 982, 1004, 1062, 1063, 1323
Iron	227, 228, 353, 982, 1004, 1062, 1323
Lead	227, 228, 230, 922, 982, 1004, 1062, 1063, 1323
Manganese	227, 228, 410, 982, 1062, 1323
Mercury	227, 228, 471, 982, 992, 1004, 1062
Molybdenum	1323
Nickel	69, 71, 227, 230, 623, 922, 982, 1323
Selenium	1062
Zinc	69, 71, 227, 228, 922, 992, 1004, 1062, 1063
Other (general)	1062, 1323
Pesticides	
Aldrin	353
DDT	1493
Synthetic/Organic Contaminants	836, 1493

TABLE 16 (continued)

Contaminants	Reference Number
Biological Contaminants	
Bacteria	79, 161, 694, 1107
Clostridium welchi	615
Coliforms	113, 568, 615, 983, 1084
Coxsackie virus (A & B)	97, 161, 890
Escherichia coli	568
Fecal streptococci	568, 615, 1084
Mycobacterium	428, 568, 615
Parasitic worms	161, 428
Polio virus	97, 99, 161, 839, 840, 890
Protozoa	428
Salmonella	161, 197, 428, 568, 615, 717
Shigella	161, 568, 615
Vibrio cholerae	161
Virus	79, 92, 95, 96, 100, 161, 352, 380, 382, 400, 428, 468, 492, 900, 1010, 1510
Other (general)	161, 531, 622, 1493

The various activated sludge processes are all able to remove over 90 percent of the soluble BOD found in wastewater. Mitchell (922) recorded data on the removals achieved with activated sludge processes of these and other water quality contaminants during his study of the Los Angeles Hyperion Treatment Plant as shown in Table 17.

TABLE 17. ACTIVATED SLUDGE TREATMENT POLLUTANT REMOVALS, LOS ANGELES, CA (922)

	Treatment Influent (mg/ℓ)	Treatment Effluent (mg/ℓ)	Percent Removal
COD	315	31	90
BOD_5	165	9	95
Suspended Solids	103	9	91
Oil and Grease	28	0.5	98
Phenols	0.09	0.009	90
Ammonia nitrogen	20	9.6	52
Phosphorus	10.1	3.3	67
Cyanide	0.30	0.13	57

As can be seen from this table, relatively high removals of most water quality contaminants can be attained in a practical application over an extended period of time. These removal efficiencies are supported by other researchers and reviewers, including Noland and Birkbeck (1003), Huang et al. (602), Rickert and Hunter (1158), Lindstedt and Bennett (799), and Besik (114).

ELEMENTAL CONTAMINANTS

Although the activated sludge process efficiently removes biodegradable organic materials, only limited removal of soluble elemental contaminants from the wastewater stream can be achieved. The removal of elemental contaminants is governed by two basic mechanisms: (1) the precipitation of metal hydroxides; and (2) the adsorption of elemental contaminants by the activated sludge floc. In either case, the elemental contaminants removed will be contained in the sludge.

When suspended solids removals were in the 90 to 95 percent range, 90 percent of the aluminum, iron, mercury, lead, and zinc settled readily with the biofloc, according to Nomura and Young (1004). Chromium (VI) and nickel median removals of 77 percent and 50 percent, respectively, were recorded under the same conditions. Morgan (969) stated that the removal percentages for chromium could vary, depending upon the treatment process used. Chromium exists in sewer systems in Cr (III) and Cr (VI); concentrations depend upon pH. Cr (III) is readily adsorbed on particles or precipitated as $Cr(OH)_3(S)$. Chromium entering as Cr (VI) experiences a strongly reducing environment (little or no dissolved O_2) in sewers and treatment plants, and is thus reduced to Cr (III) and either precipitated or adsorbed. Aeration used in activated sludge or stabilization processes can cause the resolubilization by oxidation of trivalent to hexavalent chromium, with resulting effluent water degradation.

The association of trace metals with suspended solids during the activated sludge process was investigated by Chen and Hendricks (227) and Chen and Lockwood (228) at the Hyperion Treatment Plant. Their work confirms that many trace metals are associated with suspended solids, although the concentration of trace metals on particles does not appear to depend significantly on particle size. Rather, the removal of trace metals from the waste stream by activated sludge processes depends to a great extent upon the adsorptive capability of the activated flocs.

Mitchell (922) has recorded elemental removal efficiencies for the Hyperion Plant over a three-year period, as shown in Table 18 .

TABLE 18 . REMOVALS OF TRACE METALS
BY ACTIVATED SLUDGE PROCESSES,
LOS ANGELES, CA (922)

Element	Influent (mg/ℓ)	Effluent (mg/ℓ)	Percent Removal
Copper	0.25	0.08	68
Zinc	0.42	0.23	46
Silver	0.019	0.012	37
Nickel	0.24	0.15	38
Lead	0.07	0.08	--
Arsenic	0.017	0.013	24
Cadmium	0.02	0.013	35
Chromium	0.37	0.013	96

These figures can be compared with the removal percentages as shown in Table 19 (influent metal concentrations averaged less than 1 mg/l). Clearly, the activated sludge process can reduce, but will not eliminate, trace metal concentrations in the municipal wastewater stream.

TABLE 19 . PERCENT REMOVALS OF TRACE METALS BY THE ACTIVATED SLUDGE PROCESS (1323)

Element	Average Percent Removal
Cadmium	56
Chromium	36
Copper	47
Iron	46
Lead	36
Manganese	16
Molybdenum	15
Nickel	20
Silver	48
Zinc	56

SYNTHETIC/ORGANIC CONTAMINANTS

Malaney et al. (836), in a study of the removal of possible carcinogenic organic compounds by activated sludge, concluded that no significant reduction was accomplished within normal detention times at the three treatment plants studied. The following Table 20 lists the possible carcinogens included in the analysis.

Recent work by Wachinski et al. (1439) suggests that herbicide detoxification can be achieved with a pure oxygen-activated sludge treatment system that was determined to be both economical and ecologically safe. A proprietary strain of mutant microorganisms, PHENOBAC (developed by the Worne Biochemical Corp.), was utilized that was able to degrade halogenated phenols. Even with relatively high herbicide concentrations (1380 mg/ℓ), degradation of as much as 73 percent was accomplished after a 16-day aeration period using optimum proportions of required nutrients, microflora, and oxygen. According to the authors, this figure represents a conservative estimate of possible reductions, since testing was conducted at 18°C, while the optimum growth temperature for PHENOBAC is close to 30°C.

BIOLOGICAL CONTAMINANTS

Activated sludge followed by secondary sedimentation can remove over 90 percent of coliform or pathogenic bacteria that remain after primary sedimentation; other biological pathogens are removed to varying degrees. Nonetheless, even with 90 percent removal, appreciable amounts of pathogens remain present in the effluent.

TABLE 20. POSSIBLE CARCINOGENS INCLUDED IN THE ANALYSIS (836)

2,3 - Butylene oxide
B - Propiolactone
Thiourea
Ethycarbamate
2 - Thiouracil
4 - Ethoxyphenylurea
Benzidine
4,4' - Dihydroxy-a,b-diethylstilbene
2 - Naphthylamine
4,4' - Bis (dimethylamino) benzopheuone
p-Phenylazophenol
p-Phenylazoaniline
9,10 - Dimethylanthracene
1,2 - Benzanthracene
7 - Methyl-1,2-benzanthracene
9,10 - Dimethyl-1,2-benzathracene
1,2,5,6 - Dibenzanthracene
3/4 - Benzpyrene
1,2,4,5 - Dibenzpyrene
20 - Methylcholanthrene
2 - Nitrofluorene
2 - Fluoreneamino
N-2 - Fluorenylacetamide
7,9 - Dimethylbenz (c) acridine
7,10 - Dimethylbenz (c) acridine
Dibenz (a,h) acridine
Dibenz (a,j) acridine

Pathogens can either be removed by adsorption onto the sludge flocs or destroyed by the predatory activity of the zoogleal component. A review of the literature (531, 622) reveals discussion of general wastewater removal rates with little differentiation between removals and the biocidal properties of activated sludge.

Foster and Engelbrecht (428) provided a review of pathogen removal by activated sludge processes. Conclusions of this review are summarized in Table 21 .

TABLE 21 . REMOVAL OF PATHOGENS BY THE ACTIVATED SLUDGE PROCESS (428)

Pathogen	Percent Removal
Salmonella	96 to 99
Mycobacterium	Slight to 87
Amoebic cysts	No apparent removal
Helminth ova	No apparent removal
Virus	76 to 99

Ova of intestinal parasites are apparently unaffected by the activated sludge process; in fact, the literature indicates that activated sludge-mixed liquor provides an excellent hatching medium for eggs.

A review by Hunter and Kotalik (615) provided additional pathogen removal data, as summarized in Table 22 .

TABLE 22 . PERCENT REMOVALS OF BIOLOGICAL PATHOGENS BY THE ACTIVATED SLUDGE PROCESS (615)

Pathogen	Percent Removal
Coliform	90 to 99
Fecal streptococci	84 to 94
Shigella	90 to 99
Salmonella	70
Pseudomonas aeroginosa	99
Clostridium perfringens	90 to 99
Mycobacterium tuberculosis	66 to 88

A list of the species of protozoans, nematodes, and fungi that have been found in activated sludge effluent was also presented by the authors. Species recorded include:

Protozoa -

Amoeba sp.
Euplotes patella
Loxophyllum helus
Oikomonas sp.
Pelodinium reniforme
Phyllomitus anylophagus
Trigonomonas compressar
Vorticella campanula
Epistylis pliciatils

Nematodes -

Rhabditidae
Diplogasteridae
Dorylaimidae
Monochidae

Fungi -

Alternaria
Aspergillus
Aureobasidium
Candida
Cryptococcus
Fusarium
Geotrichum
Hansenula
Kloeckera
Mucor
Penicillium
Rhodotorula
Saccharomyces
Torulopsis
Trichoderma
Trichosporon

The removal of viral contaminants by activated sludge has recently become the topic of considerable research. In general, viral removal of up to 90 percent has been observed after the activated sludge process. However, large variations in removal have been reported, probably because sampling was not temporally coordinated (468). Destruction by sewage microflora and virus adsorption to floc during the process are believed to be the main factors governing viral removal. Table 23 reports typical viral removals that can be expected from activated sludge treatment (161).

TABLE 23 . VIRAL REMOVAL BY ACTIVATED SLUDGE TREATMENT (161)

Pathogen	Percent Removal
Enteroviruses	0 to 90 percent
Polio viruses	0 to 90 percent
Coxsackie viruses	0 to 50 percent
ECHO viruses	no apparent removal

Malina et al. (839) concluded from their research that viral inactivation by activated sludge is independent of the hydraulic detention time. Polio virus adsorption to sludge is almost immediate; the adsorbed virus particles are inactivated according to first order kinetics with a half-life in the range of 4 to 5 hr. Activated sludge utilizes aeration for optimum dispersion of the flocculant masses which, along with gases produced during microbial respiration, may entrain bacterial and viral pathogens in aerosols. The aerosols released are in the 5-µm diameter range, small enough to permit lung penetration of a substantial proportion of the particles.

SECONDARY TREATMENT: TRICKLING FILTER

INTRODUCTION

Trickling filters have been widely used for secondary biological treatment of municipal wastewater, and substantial literature is available on this porcess, as indicated on Table 24 .

The trickling filter system generally consists of a tank open on both top and bottom. The tank is filled with a rock or plastic filter media having a high surface area to allow attachment of zoogleal slimes and void fraction for movement and diffusion of oxygen. Contaminant removal is accomplished through adsorption at the surface of the biological slimes covering the filter media. Following adsorption, the organics are utilized by the slimes for growth and energy. The trickling filter is followed by clarification to remove biological solids periodically flushed from the filter.

Trickling filter influent is usually from a primary treatment system. System outputs include effluent, sludge, and possible aerosols.

The literature generally refers to percent removal, with no distinction made between separation and degradation or destruction. As in the case of other secondary processes, the literature primarily addresses the general and biological contaminants and is sparse in the areas of elemental, synthetic, and biocidal contaminants.

WATER QUALITY PARAMETERS

The removal of BOD and suspended solids by trickling filters is reported to be from 65 to 95 percent, averaging about 85 percent (622, 1323). The efficiency of trickling filtration decreases as temperatures fall below 20°C. Imhoff et al. (622) reported that a reduction of temperature from 20° to 10°C results in an efficiency loss of about 40 percent. Nickerson et al. (996) found that chemical addition ahead of primary clarifiers increases overall BOD and suspended solids removals in trickling filters. Lager and Smith (745) reported that no significant removal of total nitrogen or phosphorus occurred during the conventional trickling filter process. In low-rate filters, ammonia and nitrogenous organic compounds are

TABLE 24. LITERATURE REVIEWED PERTAINING TO TRICKLING FILTERS

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	5, 41, 349, 745, 891, 1403, 1484
BOD	60, 241, 622, 745, 814, 907, 996, 1013, 1143, 1323, 1360, 1403, 1546
Chlorides	907, 1484
COD	241, 491, 891, 907, 1485
Nitrates	4, 5, 349, 745, 891, 1484
Nitrites	349, 891, 1484
Phosphates	70, 633, 745, 814, 891, 1013, 1484
Suspended solids	498, 622, 745, 996, 1013, 1323, 1403, 1546
Total organic carbon	60, 241, 615
Other (general)	391, 393, 531, 622
Elemental Contaminants	
Aluminum	814
Boron	1484
Cadmium	1323
Chromium	69, 1143, 1323
Copper	69, 1143, 1323
Germanium	1411
Iron	1323, 1484
Lead	1323, 1484
Manganese	1323, 1484
Molybdenum	1323

TABLE 24 (continued)

Contaminant	Reference Number
Nickel	69, 876, 1143, 1323
Zinc	69, 1143, 1323
Synthetic/Organic Contaminants	241, 615
Biological Contaminants	
Bacteria	79, 142, 161, 400, 486
Coliforms	486, 568, 983, 1278
Coxsackie virus (A & B)	161
Escherichia coli	568
Fecal streptococci	1278
Mycobacterium	428
Parasitic worms	161, 428, 615
Polio virus	161
Protozoa	428
Salmonella	161, 428, 717, 1278
Shigella	161
Vibrio cholerae	161, 717
Virus	79, 95, 96, 97, 100, 161, 380, 429, 492, 1278
Other (general)	161, 486, 622

usually oxidized to yield high proportions of nitrates and some nitrites in the effluent; high-rate trickling filter effluents are low in nitrates and nitrites due to limited system oxidation. Barth et al. (70) reported total phosphorous removals of up to 75 percent with direct dosing of aluminate to a trickling filter.

ELEMENTAL CONTAMINANTS

Removal of elemental contaminants by trickling filters is not well documented. A summary of the literature on metal removals as compiled by the State of California Water Resources Board is shown in Table 25.

TABLE 25. TRICKLING FILTER PROCESS REMOVAL OF TRACE METAL CONTAMINANTS (1323)

Element	Average Percent Removal
Cadmium	5
Chromium	19
Copper	47
Iron	46
Lead	36
Manganese	16
Molybdenum	15
Nickel	20
Silver	48
Zinc	56

Trace metal removals by trickling filters are substantially lower than those achieved with the activated sludge process because there is less formation and sedimentation of trace metal complexes.

BIOLOGICAL CONTAMINANTS

Trickling filters do not effectively remove many biological pathogens. Table 26 illustrates reported removal efficiencies. Organisms are adsorbed into the zoogleal slime but due to similar surface charges and morphology, biocidal effects are variable (622).

TABLE 26. REMOVAL OF PATHOGENS BY TRICKLING FILTERS (428)

Pathogen Group	Efficiency
Salmonella	88 to 99.9
Mycobacterium	66 to 99
Amoebic Cysts	11 to 99.9
Helminth Ova	62 to 76
Virus	0 to 84

Foster and Engelbrecht (428) observed that trickling filters are capable of reducing paratyphoid organisms by 84 to 99 percent. A review by Hunter and Kotalik showed 99.7 percent removal of *Schistosoma mansoni* ova. The authors also concluded that trickling filter effluents can contribute a major portion of the free living nematode population found in receiving waters.

Improperly operated low-rate trickling filters can provide an excellent breeding area for insects, especially filter flies (Psychoda) and springtails. Trickling filters cannot be depended upon to produce significant or consistent viral reductions. Foster and Engelbrecht (428) reported removals ranging from 0 to 84 percent. Berg (97) speculated that even when viruses are adsorbed, they may eventually be replaced by other substances and leach out of the filter slime as a result of an equilibrium effect.

SECONDARY TREATMENT: AERATED LAGOONS

INTRODUCTION

Aerated lagoons are ponds in which mechanical aeration is used to increase the rate at which oxygen is made available to facilitate biological stabilization. The aeration also provides mixing for suspension of microbial floc. The biological process does not include algae, and organic stablization depends on the mixed liquor that develops within the pond. Literature surveyed concerning aerated lagoons is listed in Table 27.

WATER QUALITY PARAMETERS

BOD removal by aerated lagoons is a function of aeration period, temperature, and the nature of the wastewater. The aeration of a typical domestic wastewater for five days at 20°C provides about 85 percent BOD reduction; lowering the temperature to 10°C reduces the efficiency to approximately 65 percent (531).

BIOLOGICAL CONTAMINANTS

A discussion of the literature by Parker (1052) revealed that coliform reductions in the range of 80 to 99 percent can be achieved with optimum detention time. This is supported by the experiments of Carpenter et al. (198), who reported that coliform organisms are efficiently removed by the use of aerated lagoons. Klock (717) stated that the coliform survival rate in lagoons is a function of the oxidation-reduction potential and temperature.

Berg (95) discussed the removal of viruses by stabilization ponds, concluding that virus removal can be expected to be erratic.

TABLE 27. LITERATURE REVIEWED PERTAINING TO AERATED LAGOONS

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	454
BOD	91, 484, 531, 651, 784, 1404
COD	91, 266, 484
Nitrates	454
Nitrites	454
Phosphates	328, 784
Suspended solids	484, 745
Other (general)	393, 531, 622
Biological Contaminants	
Bacteria	198, 717
Coliforms	198, 568, 717, 1052
Escherichia coli	1052
Fecal streptococci	568, 1052
Salmonella	568
Virus	95, 198, 382

SECONDARY TREATMENT: PONDING

INTRODUCTION

An oxidation or facultative pond is generally a shallow earthen basin designed to promote a symbiotic existence between algae and bacteria (745). Algal photosynthesis and surface reaction maintain aerobic conditions in the photic region, while anaerobic bacteria flourish in the aphotic zone. Ponds are normally operated in series and are sometimes used for "polishing" effluent from conventional secondary processes.

Influent to a ponding system may be raw sanitary waste, primary effluent, or secondary effluent. Pond effluent can enter the environment by direct discharge or by seepage to the groundwater. Literature reviewed concerning ponding is listed in Table 28. As can be seen from this review, only a limited amount of research has examined the removal of contaminants by the ponding process.

WATER QUALITY PARAMETERS

Oxidation pond removal efficiencies for suspended solids and BOD can vary widely and may even reach negative values (745). Removal ranges of 60 to 50 percent have been reported for suspended solids, and of 70 to 10 percent for BOD_5. This variation occurs because most influent BOD is converted into suspended algal mass. This mass exerts a BOD demand and provides suspended solids that may be carried into the effluent.

Bacterial decomposition and algal growth are both retarded by reduced temperatures, reducing removal efficiency of the ponding process (531).

SYNTHETIC/ORGANIC CONTAMINANTS

The removal of trisodium nitrilotriacetate (NTA) by ponding has been investigated by Klein (714). He found that after a two-month acclimation period, steady-state removal was in excess of 90 percent, with influent concentrations in the range of 30 mg/ℓ.

TABLE 28. LITERATURE REVIEWED PERTAINING TO PONDING

Contaminant	Reference Number
Water Quality Parameters	
BOD	326, 531, 745
COD	1484
Chlorides	427
Fluorides	427
Nitrates	226, 326, 375, 446, 1432
Phosphates	326, 375, 426, 446, 1484, 1500
Suspended solids	393, 745
Other (general)	393, 531, 745
Synthetic/Organic Contaminants	714
Biological Contaminants	
Bacteria	842, 1278
Coliforms	849, 850, 1278
Escherichia coli	850, 1048
Fecal streptococci	849, 850, 1278
Salmonella	669
Virus	95, 96, 97, 1278

BIOLOGICAL CONTAMINANTS

Ponding preceded by adequate secondary treatment provides excellent bacteria removal. Kampelmacher and Jansen (669) found that removal of salmonella by oxidation ponds was not inferior to removal achieved by conventional treatment plants. Species of the coliform group, although reduced by ponding, are not effectively eliminated according to Parhad and Rao (1048). Slanetz et al. (1278), however, reported that if two ponds were operated in a series at temperatures of 17° to 26°C, the die-off rate of coliform, fecal coliforms, and fecal streptococci ranged from 95 to 99 percent. During winter when temperatures were in the 1° to 10°C range, the die-off rate was 46 times lower. Berg (97) states that virus removal by ponding is erratic, ranging from 0 to 96 percent; virus recovery decreased as the effluent passed through a series of maturation lagoons.

TERTIARY TREATMENT: CHEMICAL TREATMENT

INTRODUCTION

The purpose of chemical treatment is to coagulate suspended solids and precipitate phosphate and various trace metals. Chemical coagulation of secondary effluents may be accomplished by the addition of lime, alum, polymers, or iron salts, and involves three operations: (1) injection and rapid mixing of the coagulants to neutralize the predominantly negative charges on suspended matter; (2) gentle stirring to promote agglomeration of the coagulated particles into large, settleable floc; and (3) sedimentation to provide gravity separation of the flocculated material from the waste water. The settled material is transferred to a sludge handling system. As indicated by Table 29, a great deal of information is available concerning the removal of various public health impairing contaminants by chemical treatment processes.

WATER QUALITY PARAMETERS

Culp and Shuckrow (278) investigated chemical treatment of raw wastewater with lime and found that removals of 95 to 98 percent phosphorus and 99 percent suspended solids can be achieved with chemical clarification followed by carbon adsorption. The treatment of municipal wastewater with alum precipitation as studied by Shuckrow et al. (1254) resulted in removal efficiencies of 85 percent for COD and 83 percent for total organic carbon.

The removal of BOD, suspended solids, and phosphorus as reviewed by Lager and Smith (745) is summarized in Table 30 . Removals from secondary effluents of the magnitudes listed obviously provide a high quality effluent.

TABLE 29 . LITERATURE REVIEWED PERTAINING TO CHEMICAL TREATMENT

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	43, 120, 277, 284, 390, 601, 614, 885, 891, 1321, 1510
BOD	119, 124, 277, 278, 284, 334, 498, 614, 745, 794, 799, 813, 814, 878, 885, 996, 1012, 1013, 1215, 1337, 1388, 1393, 1402, 1467, 1510
Chlorides	885
COD	119, 120, 124, 249, 277, 278, 284, 329, 498, 885, 1254
Cyanides	107, 922, 993
Fluorides	410, 1321
Nitrates	119, 120, 124, 249, 277, 278, 284, 329, 498, 885
Nitrites	284, 390, 891, 1085, 1321, 1510
Phosphates	278, 284, 290, 295, 326, 328, 375, 410, 423, 479, 513, 601, 614, 650, 711, 743, 745, 763, 794, 799, 814, 865, 876, 885, 891, 1012, 1013, 1036, 1039, 1213, 1215, 1265, 1311, 1321, 1337, 1341, 1393, 1402, 1466, 1500, 1510, 1547, 1552
Suspended solids	38, 108, 119, 120, 249, 278, 330, 498, 601, 701, 745, 799, 813, 896, 996, 1012, 1013, 1215, 1254, 1338, 1389, 1393, 1402, 1485, 1510
Total dissolved solids	277, 1215, 1321
Total organic carbon	119, 120, 124, 284, 1254, 1467
Other (general)	385, 391, 393, 531, 622

TABLE 29 (continued)

Contaminant	Reference Number
Elemental Contaminants	
Aluminum	334, 814, 1036, 1383
Antimony	38
Arsenic	38, 855, 922, 1062, 1274, 1323, 1383
Barium	38, 855, 1062, 1323, 1383
Boron	1383
Cadmium	38, 39, 107, 802, 855, 922, 1103, 1323, 1383, 1504, 1505, 1506
Chromium	38, 39, 204, 802, 855, 1323, 1385, 1504, 1505, 1506
Cobalt	1383
Copper	38, 39, 107, 204, 855, 922, 1323, 1383, 1504, 1505, 1506
Iron	38, 334, 763, 1323, 1383, 1504, 1506
Lead	855, 922, 1323, 1383, 1411, 1504, 1506
Manganese	38, 334, 410, 855, 1323, 1383, 1504, 1505, 1506
Mercury	38, 775, 811, 855, 922, 1077, 1323, 1353, 1383, 1504, 1505, 1506
Molybdenum	38, 1323
Nickel	38, 855, 922, 1323, 1383, 1504, 1505
Selenium	802, 1323, 1383
Uranium	38, 1323
Zinc	38, 107, 855, 922, 1323, 1383, 1504, 1505, 1506
Other (general)	38, 760, 799, 1323, 1506

TABLE 29 (continued)

Contaminant	Reference Number
Synthetic/Organic Contaminants	6, 1496
Biological Contaminants	
Adeno virus	1522
Bacteria	79, 331, 335, 670, 799, 830, 891
Coliforms	101, 284, 469, 799, 896, 1393
Coxsackie virus (A & B)	97, 1522
ECHO virus	1522
Escherichia coli	880, 1183, 1382
Fecal streptococci	799
Hepatitis virus	469, 1522
Parasitic worms	830
Polio virus	103, 159, 223, 499, 837, 1293, 1313, 1382, 1510, 1522
Protozoa	830
Salmonella	670
Virus	79, 92, 96, 97, 159, 219, 220, 245, 246, 259, 283, 284, 331, 380, 381, 468, 848, 1009, 1057, 1071, 1243, 1302, 1312, 1313, 1510, 1511

TABLE 30. REMOVALS ACHIEVED BY CHEMICAL CLARIFICATION (745)

Chemical	BOD Removal (Percent)	Suspended Solids Removal (Percent)
Lime pH 11.5	80	90
Ferric Chloride 170 mg/ℓ dose	80	95
Ferric Chloride 80-100 mg/ℓ dose	75	--

Phosphate removal by chemical precipitation (lime) has received considerable attention in the literature in recent years. The research on phosphate removal in general indicates that lime clarification usually provides removal efficiencies greater than 90 percent. This is supported by the work of Davis (295), Sturm and Hatch (1341), Johnson (650), and Bernhardt et al. (108).

ELEMENTAL CONTAMINANTS

The precipitation of metal hydroxides from solution is governed by the pH and the concentration of the metal ion in solution. Since many of the trace metals form insoluble hydroxides near pH 11, lime coagulation results in a reduction of these metal concentrations. Table 31 (1323) summarizes the effects of lime coagulation on a number of heavy metals. Some of the data were collected on industrial metal wastes characterized by metal ion concentrations a great deal higher than would occur in any municipal plant influent. The data are included here as examples of possible metal reductions, since since such figures from chemical coagulation of municipal wastewaters are scarce.

As can be seen from these figures, arsenic, molybdenum, and selenium had relatively poor removal rates,and the potential removal of mercury was estimated to be low. Only 11 percent of hexavalent chromium was removed, although the trivalent form was reduced more than 99.9 percent. Most other metals tested were very effectively reduced at high pH. Lower removals of these same metals (usually less than 50 percent) can be achieved with alum coagulation at near neutral pH values (1323), a fact that illustrates the dependence of precipitation on pH.

Gulledge and O'Connor (520) studied the removal of arsenic (V) from tap water in jar tests using alum and ferric sulfate.

TABLE 31 . REMOVAL OF ELEMENTAL CONTAMINANTS BY LIME COAGULATION (1323)

Metal	Concentration Before Treatment mg/ℓ	Concentration After Treatment mg/ℓ	Percent Removal
Antimony[a]	--	--	90
Arsenic[a]	--	--	<10
	23	23	0
Barium[a]	--	1.3 (sol)[b]	
Bismuth[a]	--	.0002 (sol)	
Cadmium	Trace	--	Abt. 50
	0.0137	0.00075	94.5
Chromium (+6)	0.56	0.050	11
Chromium (+3)	7,400	2.7	99.9+
	15	0.4	97
Copper	15,700	0.79	99.9+
	7	1	86
	7	.05	93
	302	Trace	99+
	15	0.6	97
Gold[a]	--	<.001 (sol)	90+
Iron	13	2.4	82
	17	0.1	99+
	2.0	1.2[c]	40
Lead[a]	--	<.001 (sol)[b]	90+
	15	0.5	97
Manganese	2.3	<0.1	96
	2.0	1.1[c]	45
	21.0	0.05	95
Mercury[a]	--	Oxide Soluble	<10
Molybdenum	Trace	--	Abt. 10
	11	9	18

TABLE 31 (continued)

Metal	Concentration Before Treatment mg/ℓ	Concentration After Treatment mg/ℓ	% Removal
Nickel	160	0.08	99.9+
	5	0.5	90
	5	0.5	90
	100	1.5	99
	16	1.4	91
Selenium	0.0123	0.0103	16.2
Silver	0.0546	0.0164	97
Telurium[a,d]	--	(<0.001?)	(90+)
Titanium[a,d]	--	(<0.001?)	(90+)
Uranium[e]	--	?	?
Zinc	17	0.3	98
	--	.007 (sol)	90+

[a] The potential removal of these metals was estimated from solubility data.

[b] Barium and lead reductions and solubilities are based upon the carbonate.

[c] These data were from experiments using iron and manganese in the organic form.

[d] Titanium and telurium solubility and stability data made the potential reduction estimate unsure.

[e] Uranium forms complexes with carbonate ion. Quantitative data were unavailable to allow determination of this effect.

In each case, increased coagulant dosage resulted in increased arsenate removal. Dependence on pH was again seen in this study. Under optimum conditions, removals of 85.8 percent with alum and 96.8 percent with ferrous sulfate were achieved.

Pilot plant studies of municipal wastewater containing 5 mg/ℓ arsenic cited by Patterson (1062) suggest that chemical treatment can provide efficient removal of this element. Ferric sulfate at 45 mg/ℓ Fe and pH 6.0 removed 90 percent of the arsenic; lime at 600 mg/ℓ and pH 11.5 removed 73 percent. In similar studies cited by Patterson, barium removals of 97 percent were obtained when municipal wastewater dosed with 5 mg/ℓ barium was treated with 45 mg/ℓ Fe at pH 6.0. Lime at 600 mg/ℓ and pH 11.5 resulted in 80 percent removal. The removal of cadmium from waters by sorption onto hydrous oxides of solid metals such as manganese and iron was investigated by Posselt and Weber (1103). They concluded that sorptive uptake of cadmium on such materials would constitute a method easily adaptable to present treatment technology.

BIOLOGICAL CONTAMINANTS

Chemical treatment can be used to reduce or remove many biological pathogens present in municipal wastewater. Lindstedt and Bennett (799) evaluated the effectiveness of lime clarification in reducing bacterial concentrations, finding that treatment effectiveness increases with increasing chemical dosage and pH. At a lime dosage of 400 mg/ℓ , fecal coliform, fecal streptococcus, and total coliform concentrations could be reduced by two orders of magnitude. It was also found that about 90 percent removal of bacteria can be achieved through alum clarification over a broad range of alum dosage.

Jar tests employing the f2 bacteriophage virus, lake water, and a variety of chemical coagulants and polyelectrolyte coagulant aids were conducted by York and Drewry (1522). As shown in Table 32 , aluminum sulfate (alum), ferric chloride, ferric sulfate, ferrous sulfate, and polyelectrolyte B were found to give maximum virus removals greater than 90 percent at optimum dosage.

TABLE 32 . COMPARISON OF THE EFFECTIVENESS OF THE COAGULANTS TESTED (1522)

Coagulants-Coagulant Aids	Dose mg/ℓ	Maximum Virus Removal percent
$Al_2(SO_4)_3$	25	99.9
$FeCl_3$	50	99.4
$Fe_2(SO_4)_3$ x H_2O	50	92.0
$FeSO_4$ and $Ca(OH)_2$	36	93.5
$Al_2(SO_4)_3$ and $Na_2OAl_2O_3$	30 23	98.6
Polyelectrolyte A	2.0	76
$Al_2(SO_4)_3$ and polyelectrolyte B	18 1.0	99.2
Polyelectrolyte B	2.0	99.6
$Al_2(SO_4)_3$ and polyelectrolyte B	18 0.7	99.8
$Al_2(SO_4)_3$ and polyelectrolyte C	18 0.4	99.3
$Al_2(SO_4)_3$ and polyelectrolyte E	18 0.1	99.3
$Al_2(SO_4)_3$ and polyelectrolyte F	18 0.1	99.6
$Al_2(SO_4)_3$ and polyelectrolyte D	18 1.0	99.4

Berg et al. (103) experimentally mixed polio virus I and secondary effluent in containers, added lime, and stirred the solution for 15 min to allow formation of floc particles. Settling followed for 60 to 75 min. The removals obtained with varying dosages of lime are shown in Table 33 . Large coagulant doses were capable of effecting virus removals of up to 99.9 percent. After further investigation, the authors concluded that significant destruction of viruses can be attributed to the high pH occurring with high lime concentrations (in the range of 400 to 500 mg/ℓ).

TABLE 33 . REMOVAL OF POLIO VIRUS I FROM SECONDARY EFFLUENT BY FLOCCULATION WITH $Ca(OH)_2$ (103)

$Ca(OH)_2$ Concentration mg/ℓ	Initial Virus Concentration pfu/ℓ	pH of Treated Effluent	Surviving Virus Concentration pfu/ℓ	Virus Removal percent
200	33,333	9.30	2,200	92.3
300	51,480	10.21	15,912	69.1
400	55,000	11.30	1,940	96.5
500	33,333	11.03	505	98.5
500	33,333	11.01	47	99.86

Chaudhuri and Engelbrecht (219) used water devoid of extraneous organic matter in their laboratory investigation of virus removal with alum. Using the experimentally determined optimum pH and dosage, 98.0 percent removal of bacteriophage T4 and 99.9 percent removal of bacteriophage MS2 were obtainable. However, the addition of organic matter in the form of albumin or treated wastewater lowered these efficiencies. For example, only 95.7 percent removal of bacteriophage T4 was obtained after the addition of 200 mℓ/ℓ of settled wastewater. Further experiments demonstrated no inactivation of the virus particles that were removed in the settled floc. However, Brunner and Sproul (159) demonstrated 60 percent permanent inactivation in the case of polio virus I removed from solution with aluminum phosphate precipitates. Their studies of polio virus I and bacteriophage T2 removal showed that under optimum conditions, removals can reach 98 and 94 percent, respectively, with aluminum and calcium precipitation. Actual removals are related to pH and chemical dosage.

Chemical treatment (high pH) holds considerable promise as a means of effectively inactivating or destroying pathogenic organisms contained in wastewater. By itself, chemical treatment cannot be relied upon to produce a pathogen-free effluent; used in conjunction with disinfection, however, it can help ensure that such an effluent is achieved.

TERTIARY TREATMENT: FILTRATION

INTRODUCTION

Inability of gravity sedimentation in secondary clarifiers to remove small particles (and associated public health impairing contaminants) is a limitation of BOD and suspended solids removal by conventional wastewater treatment. Filtration as a tertiary process upgrades treatment performance by removing a portion of the unsettled suspended solids from secondary effluents. In addition, filtration often precedes other tertiary processes such as adsorption and ion exchange since the presence of suspended solids interferes with the operation of these processes. Reviewed literature on filtration is shown in Table 34.

Filtration of wastewater to reduce the suspended solids concentration is accomplished by passage through a bed of granular particles. Single, dual, or mixed media beds may be used, composed of anthracite coal, granite, sand, and/or gravel (531). Suspended solids are removed by a variety of mechanisms: straining, impingement, settling, and adhesion. The treatment efficiency of the process is influenced by:

- The concentration and characteristics of the wastewater solids (particle-size distribution, surface characteristics, organic versus inorganic, etc.)
- The characteristics of the filter media and filtering aids used (particle-size distribution, surface characteristics, etc.)
- The design and operation of the filter.

Since wastewater flow rate and solids content are variable, and processes upstream of filtration may vary in performance, the efficiency of filtration may also be expected to vary. For this reason, values presented in the following discussion should be considered as merely indicative of the range of achievable removals.

WATER QUALITY PARAMETERS

In general, the best effluent quality achievable by plain filtration of secondary effluent is about 5 to 10 mg/ℓ for suspended solids and BOD. The suspended solids content of

TABLE 34. LITERATURE REVIEWED PERTAINING TO FILTRATION

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	43, 548, 885, 1323, 1546
BOD	119, 314, 697, 771, 885, 1319, 1323, 1393, 1546
COD	119, 771, 885, 1319, 1323
Chlorides	114, 885, 1062, 1319, 1367
Cyanides	1062
Fluorides	1062
Nitrates	384, 885, 1319, 1367
Oil and grease	1062
Phosphates	108, 119, 314, 659, 885, 1323, 1393
Suspended solids	39, 108, 454, 531, 659, 771, 1323, 1367, 1393, 1452, 1546
Total dissolved solids	1062
Total organic carbon	114, 119, 1323
Other (general)	385, 531, 622
Elemental Contaminants	
Arsenic	1062, 1244
Barium	1062
Boron	1062
Cadmium	38, 39, 802, 1062, 1063, 1559
Chromium	38, 39, 802, 1062, 1559
Copper	38, 39, 697, 1062, 1063, 1319, 1559
Iron	38, 1062, 1319, 1559

TABLE 34 (continued)

Contaminant	Reference Number
Lead	1062, 1063, 1559
Manganese	38, 635, 1062, 1319, 1559
Mercury	1062
Nickel	38, 1062, 1063
Selenium	38, 802, 1062
Zinc	38, 697, 1062, 1063, 1319, 1559
Other	38, 1062
Synthetic/Organic Contaminants	1062, 1496
Biological Contaminants	
Bacteria	161, 331
Coliforms	101, 469, 608, 617
Coxsackie virus (A & B)	95, 161
ECHO virus	1182
Hepatitis virus	469
Parasitic worms	161, 1469
Polio virus	103, 153, 161, 223, 1168, 1182, 1293
Salmonella	161
Shigella	161
Vibrio cholerae	161, 424
Virus	25, 95, 96, 152, 154, 161, 261, 331, 454, 468, 492, 608, 1009, 1240, 1312
Other (general)	153, 161

secondary effluent was reduced to 5 mg/ℓ with both rapid sand and mixed media filters, employed respectively at a treatment and a pilot plant. Complete removal, however, could not be effected. If further reduction is desired, chemical coagulation must precede filtration (454, 531).

Two studies at a pilot facility and a major treatment plant (1323) indicated that essentially complete suspended solids removal was accomplished when filtration was preceded by chemical treatment of secondary effluent. Chemical clarifier effluent contained 0.7 mg/ℓ total phosphorus; after filtration, this phosphorus content was reduced to 0.1 mg/ℓ. At the pilot facility, the filter effluent contained 17.6 mg/ℓ COD, 9 mg/ℓ total organic carbon (TOC), and no detectable phosphorus, compared with filter influent concentrations of 18.1 mg/ℓ COD, 8.6 mg/ℓ TOC, and 0.4 mg/ℓ phosphorus.

When chemically treated secondary effluent was applied to rapid-sand filters, generally 20 to 60 percent of applied BOD was removed, 30 to 70 percent of phosphate, and 40 to 80 percent of the suspended solids (1393). These values were obtained using a variety of influent concentrations and chemical dosages. The results of 30 days continuous operation of a pilot plant practicing filtration preceded by chemical treatment of secondary effluent are summarized in Table 35 (1323).

TABLE 35. RESULTS OF ONTARIO, CANADA, PILOT PLANT STUDY INVOLVING FILTRATION PRECEDED BY CHEMICAL TREATMENT OF SECONDARY EFFLUENT (1323)

Quality Parameter	Raw Wastewater	Secondary Effluent	Filter Effluent
Total Organic Carbon (mg/ℓ)	110-165	14-28	4.5-7.5
BOD_5 (mg/ℓ)	230-400	5-14	2.0-3.0
PO_4 (as PO_4)	9-21	1.3-3.5	0.4-1.0
Total Nitrogen N	27-51	25-37	20-35
Ammonia N	17-29	21-29	18-29
Suspended Solids (mg/ℓ)	148-268	13-37	3-12

Results of a pilot plant study at Cleveland, Ohio - where chemical coagulation and settling of raw wastewater was followed by filtration and granular carbon adsorption - were reviewed by Culp and Shuckrow (278). On the basis of the data provided, removals attributable to chemical treatment and filtration can be calculated to be 66 percent of the applied BOD and 77 percent of the applied COD. A second pilot plant study reviewed by the authors involved the same treatment scheme. Removals at this plant due to combined chemical treatment and filtration can be inferred to range from 77 to 84 percent.

While studying soil filtration, de Vries (314) applied primary effluent to a sand filter and obtained BOD and phosphate removals of nearly 100 percent. Phosphate removals were attributed to the natural coatings of Fe_2O_3 and Al_2O_3 on sand grains.

ELEMENTAL CONTAMINANTS

After chemical treatment, filtration to remove residual particulate matter may provide some additional removal of elemental contaminants. Elemental contaminant removals achieved by filtration depend primarily upon the extent of suspended solids removals, with which the various trace elements are associated. Table 36 , compiled from the literature by Argo and Culp (38) gives results for sand filtration of some municipal and industrial wastes.

TABLE 36 . HEAVY METAL REMOVAL BY SAND FILTRATION FOLLOWING LIME COAGULATION (38)

Metal	Concentration Before Filt.	Concentration After Treat.	pH	% Removal By Filt.
Cd	Trace to		8.1	95
	0.00075 mg/ℓ	0.00070	7.6	6.6
Cr^{+6}	0.0503	0.049	7.6	2.6
Cr^{+3}	2.7	0.63	8.7	77
Cu	0.79	0.32	9.5	59.5
	-	.5		
Fe	-	0.1	10.8	
		1.2 Organic	10.5	
Mn	-	0.1	10.8	
		1.1 Organic	10.5	
Ni	0.08	0.1	8.7	
		0.5	9.5	
Se	.0103	0.00932	11	9.5

TABLE 36 (continued)

Metal	Concentration Before Filt.	Concentration After Treat.	pH	% Removal By Filt.
Ag	0.00164	0.00145	11	11.6
Zn	0.97	0.23 2.5	8.7 9.5	76.3

Patterson (1062) cites evidence from pilot plant studies that little or no additional removal of arsenic was afforded by filtration of chemically treated municipal wastewater.

BIOLOGICAL CONTAMINANTS

Several investigations have been reported concerning the removal of viruses by sand filtration. It has been shown that insignificant virus removal is achieved by rapid filtration through clean sand (25, 95). However, virus removal efficiency will be increased by impregnation of the filter medium with coagulated floc, the presence of organic matter trapped in the sand, chemical flocculation prior to filtration, or a decrease in the filtration rate. The addition of iron salts prior to filtration has resulted in significantly higher coliform reductions, as discussed by Hunter et al. (617). Similarly, Robeck et al. (1168) noted that if a low dose of alum was fed to a rapid coal and sand filter just ahead of filtration, more than 98 percent of polio virus Type I could be removed. If the dosage was increased and conventional flocculators and settling were used, removal was increased to over 99 percent. The authors also noted a general trend toward better removal of polio virus I with slower filtration rates although their data were often erratic. At slow sand filter rates (0.6 to 1.2 ℓ/min/m^2), removal ranged from 50 percent to about 98 percent. At rapid filtration rates (38 to 76 ℓ/min/m^2), virus removals ranged from about 10 to 70 percent. Similarly, research examined by Berg (95) showed that filtration through sand at 7.5 ℓ/min/m^2 removed 99 percent of the coxsackie virus A5 while filtration at 75 ℓ/min/m^2 removed only about 10 percent of the virus.

Brown et al. (153) reported 70 to 90 percent removals of low concentrations of either bacteriophage T2 or polio virus Type I by filtration through uncoated diatomaceous earth. However, no significant virus removals by uncoated diatomaceous earth were achieved in a laboratory study by Amirhor and Engelbrecht (25). With polyelectrolyte-coated filter media, removals greater than 99 percent were consistently achieved (25).

In laboratory tests by Berg et al. (103), from 82 to greater than 99.8 percent of polio virus I was removed from chemically treated effluent by rapid sand filtration. The results of these tests are given in Table 37 .

TABLE 37 . REMOVAL OF POLIO VIRUS I FROM $Ca(OH)_2$ FLOCCULATED EFFLUENT BY RAPID SAND FILTRATION AS MEASURED BY MEMBRANE FILTER RECOVERY OF VIRUS (103)*

Test No.	Virus Concentration pfu/ℓ		Virus Removal percent
	Before Sand Filtration †	After Sand Filtration	
1	2,200	397	82.0
2	15,912	750	95.3
3	1,940	<4.6	>99.8
4	505	12.5	97.5
5	47	2.8	94.0

* Filtration rate 2.25 gpm/sq ft through 8 in of sand.

† Virus concentration in flocculated effluent just prior to sand filtration.

Laboratory experiments on the removal of nematodes by rapid sand filtration were conducted by Wei et al. (1469). Removal efficiency was about 96 percent when all the nematodes in the influent were dead or nonmotile. However, most motile nematodes were able to penetrate the filter bed.

Sand filtration may also provide some removal of amoebic cysts and ascaris eggs, according to a literature survey by Bryan (161). He did not indicate the levels of removal afforded.

TERTIARY TREATMENT: ADSORPTION

INTRODUCTION

Adsorption refers to the removal from water or wastewater streams of dissolved contaminants by their attraction to and accumulation on the surface of an adsorbent substance. Activated carbon is the most widely used adsorbent in municipal wastewater treatment to remove trace organics. Adsorption using activated carbon is utilized as a tertiary treatment step, usually following sand or multimedia filtration.

Carbon adsorption systems generally utilize granular or powdered activated carbon packed in a column or forming a filter bed through which wastewater is passed. Three consecutive steps occur in the adsorption of wastewater contaminants by activated carbon: (1) the film diffusion phenomenon, or the transport of the adsorbate through a surface film surrounding the activated carbon; (2) the diffusion of the adsorbate within the pores of the activated carbon; and (3) adsorption on the interior surfaces of the activated carbon.

Carbon adsorption of contaminants has been the topic of many research projects as shown in the literature surveyed for this report, tabulated in Table 38 .

WATER QUALITY PARAMETERS

Adsorption is most effective for removing refractory and other organics from wastewater. This is especially important when effluents of exceptional quality are required (e.g., for groundwater recharge or other reuse applications). Adsorption can be used either as a polishing step or as the major treatment process (1467). Rizzo and Schade (1165) and Zanitsch and Morand (1554) reported that carbon columns alone were capable of about 85 percent removal of BOD from wastewaters entering the columns. Bishop et al. (119, 123) and Zanitsch and Morand (1554) reported 75 to 80 percent TOC removals under the same conditions. Weber et al. (1467) found that a treatment system composed of primary settling, ferric chloride coagulation, and carbon adsorption could remove up to 97 percent of the influent BOD.

There is some disagreement among researchers regarding the ability of activated carbon to remove nitrogen species from

TABLE 38. LITERATURE REVIEWED PERTAINING TO ADSORPTION

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	113, 119, 120, 196, 885
BOD	113, 119, 120, 156, 170, 196, 278, 315, 326, 498, 522, 745, 885, 1165, 1467, 1554
COD	119, 120, 156, 196, 278, 329, 498, 668, 885, 1165, 1547
Chlorides	114, 885, 1026, 1289
Nitrates	113, 119, 120, 326, 1467
Oil and grease	480
Phosphates	24, 113, 114, 119, 120, 196, 326, 448, 885, 1467
Suspended solids	39, 113, 119, 120, 170, 196, 278, 326, 498, 668, 745, 1165, 1442
Total dissolved solids	196, 454, 1547
Total organic carbon	113, 114, 119, 120, 123, 315, 522, 1086, 1165, 1262, 1467, 1554
Other (general)	531, 622
Elemental Contaminants	
Aluminum	1383
Arsenic	855, 1383
Barium	855, 1383
Boron	1140, 1383
Cadmium	38, 39, 619, 753, 760, 855, 1383
Chromium	38, 39, 603, 619, 753, 760, 855, 1169, 1383
Cobalt	753, 1383

TABLE 38 (continued)

Contaminant	Reference Number
Copper	39, 620, 855, 1383
Iron	620, 753, 1383
Lead	753, 855, 1383
Manganese	753, 855, 1873
Mercury	753, 811, 855
Nickel	753, 855, 1383
Selenium	38, 1062, 1383
Zinc	620, 855, 1383
Other (general)	38, 760
Biocidal Contaminants	
Chlorinated hydrocarbons	762, 1221
Dieldrin	523
Herbicides	514, 516, 1221
Other (general)	523, 628, 762, 1323
Synthetic/Organic Contaminants	123, 315, 454, 480, 523, 628, 651, 762, 862, 1153, 1290, 1323, 1465, 1496
Biological Contaminants	
Bacteria	283
Coliforms	113
Escherichia coli	263
Polio virus	223, 466, 467, 1293, 1313
Virus	246, 263, 468, 492, 1010, 1312, 1313

wastewaters. Bishop et al. (119) reported that carbon adsorption had little effect on nitrogen concentrations. On the other hand, Weber et al. (1467) found that their primary settling ferric chloride coagulation, carbon adsorption system removed 95 percent of the influent nitrate; the reduction was attributed partly to biological populations growing in the columns.

Weber et al. (1467) reported that their pilot system also removed 90 percent of the phosphate in the influent wastewater. However, much of the research on phosphate adsorption has been concerned with adsorbents other than carbon. Gangoli and Thodos (448) reported that both Fl Alumina and fly ash were capable of removing up to 99 percent of influent phosphate levels. Ames and Dean (24) demonstrated that an alumina column could treat up to 400 column volumes before reaching the 10 percent phosphorus breakthrough level.

Since color is contributed largely by organic compounds in water, high color removal levels with adsorption should be possible. Zanitsch and Morand (1554) demonstrated 90 percent color removal. They also noted an 86 percent suspended solids removal, presumably by filtration.

ELEMENTAL CONTAMINANTS

Not a great deal of literature is available on the removal of elemental contaminants by carbon adsorption. Such systems are not specifically designed to remove ionic elemental contaminants, but some elementals are incidentally removed. When the metallic contaminants are in an organometallic complex, carbon adsorption columns can remove specific species. Literature from several sources (619, 760) reveals that high removals (95 percent) of cadmium and hexavalent chromium by carbon adsorption are possible. Huang and Wu (603) found that the efficiency of chromium removal increases with decreasing chromium concentration. While the mechanism of removal is not well understood, Roersma et al. (1169) were able to describe the reduction of Cr^{+6} to Cr^{+3}. The Cr^{+6} is adsorbed within the pores of the activated carbon which, in turn, is slowly oxidized to CO_2, reducing the chromium ion.

Activated carbon treatment of a secondary-treated municipal wastewater was found to reduce selenium from 9.32 to 5.85 ppb in a study cited by Patterson (1062). This represents a 37 percent removal efficiency. Patterson cited a second study in which the selenium removal efficiencies of several advanced wastewater treatment processes were determined in the treatment of secondary effluent containing 2.3 ppb selenium. While sand filtration alone reduced the effluent concentration by 9.5 percent, sand filtration followed by activated carbon treatment provided a cumulative removal of 43.2 percent of the selenium from the secondary effluent.

Logsden and Symons (811) researched the removal of mercury by carbon adsorption and found that powdered carbon, in a jar test, would adsorb both inorganic and methyl forms in excess of 70 percent.

BIOCIDAL CONTAMINANTS

Carbon adsorption is widely applied to remove organic or metal-organic biocides. The removal of insecticides and pesticides has been reviewed by Hager and Flentje (523). Dieldrin, lindane, parathion, and 2, 4, 5-T ester were reduced below the detectable limit of 0.01 ppb with influent concentrations of 3.6 to 11.4 ppb. Influent concentrations of 3, 5 dinitro-o-cresol of 30 to 180 ppb were reduced to less than 1 ppb by carbon adsorption. It was concluded that granular carbon beds will provide a margin of safety for treatment of water containing varying insecticide or pesticide residues.

Activated carbon removals of several pesticides and PCB's are well illustrated by results of laboratory studies cited by the California State Water Resources Control Board (1323), as shown in Table 39 . A variety of pesticides were experimentally added to distilled water and passed through carbon filters to test removal efficiencies. Schwarz (1221) investigated the adsorption of isoprophyl N-(3-chlorophenyl) carbamate (CIPC) onto activated carbon, concluding that powdered activated carbon readily adsorbs CIPC from aqueous solution. The adsorption of CIPC on activated carbon was independent of the pH in the range of 5 to 9.

Grover and Smith (516) studied adsorption onto activated carbon of the acid and dimethylamine forms of 2, 4-D, and dicambamate. A strong adsorption effect was noted on both the acidic and salt forms of the compounds. This effect was expected to increase at low pH values.

On the basis of the literature which has been reviewed, it can be concluded that activated carbon adsorption is effective in the removal of some biocidal contaminants; however, further investigation of this process will be useful.

SYNTHETIC/ORGANIC CONTAMINANTS

As was mentioned earlier, activated carbon and many synthetic compounds are effective at removing organic contaminants from aqueous solutions, particularly organics of low water solubility. In general, carbon adsorption following secondary treatment is capable of producing an effluent with from 1 to 7 mg/ℓ of organic carbon (123, 454).

TABLE 39. REMOVAL OF SPECIFIC TOXIC MATERIALS BY CARBON ADSORPTION (1323)

Carbon Dosage (mg/ℓ)	Residual (ppb)								
	Aldrin	Endrin	Dieldrin	DDT	DDD	DDE	Toxaphene	Arochlor 1242	Arochlor 1254
Control	48	62	19	41	56	38	155	45	49
1.0	--	--	--	41	--	34	147	--	--
2.0	26	15	6.3	--	6.9	--	80	7.3	37
2.5	--	--	--	21	--	29	--	--	--
5.0	15	3.4	2.4	3.7	3.7	12	31	1.6	17
10.0	12	1.5	1.1	--	2.2	--	2.7	1.1	4.2
12.5	--	--	--	<1	--	3.3	--	--	--
25.0	6.3	0.56	--	--	0.45	1.1	--	--	1.6
50.0	4.4	0.22	--	--	0.35	0.9	--	--	1.2

Bishop et al. (123) found that carbon adsorption was least effective in removing highly polar, highly soluble organic species. DeWalle and Chian (315) found that removal was a function, at least in part, of the molecular weight of the synthetic organic contaminant: low (<100) and high (>50,000) molecular weight compounds are poorly adsorbed. The major fraction removed by adsorption falls into a 100 to 10,000 molecular weight range.

Studies of the adsorption of 93 petrochemicals by Giusti et al. (480) confirmed the results of Bishop et al. (123) and DeWalle and Chian (315): adsorption is largely a function of molecular weight, polarity, solubility, and branching. Function was seen to have a substantial effect in conjunction with solubility and polarity. The relative amenabilities to carbon adsorption of straight chain molecules of fewer than four carbon atoms were as follows: aldehydes >esters >ketones >alcohols >glycols. For larger molecules, the alcohols moved ahead of the esters.

Much of the research done on the adsorption of synthetic organic compounds from wastewater has been concerned with determining mechanisms of adsorption and optimum removal conditions. The only general removal efficiency studies available report removals in terms of total organic carbon with little or no effort made to differentiate the organic compounds involved. Based on these results, adsorption can reduce the levels of synthetic organic compounds in a typical domestic wastewater by 75 to 85 percent. If a particular type or types of organic compounds predominate in a wastewater, these removals must be adjusted to reflect the effect of compound character on the adsorption process.

BIOLOGICAL CONTAMINANTS

With the exception of enteroviruses, no information was found on the adsorption of biological contaminants, although incidental removal of other organisms would be expected by filtering action. Adsorption brings about simple removal of viruses from wastewater rather than inactivation or destruction (263, 468). Consequently, viable viruses could be reintroduced to wastewater should desorption of viruses adsorbed to activated carbon occur.

Columns of granular activated carbon removed between 18 and 40 percent of Type I polio virus from secondary effluent in studies by Sproul et al. (1313). This research and research by Gerba et al. (467) using Type I polio virus indicate that adsorption is inversely related to the concentration of organic matter in the wastewater. The organics and the virus compete for adsorption sites; consequently, desorption of virus can occur as adsorption of organic matter continues, or if the concentration of organics is increased. Several authors have thus concluded that the process is not dependable for producing virus-free effluents (246, 468, 1313).

The level of removal actually attained is closely related to the type of treatment that precedes adsorption. For example, reducing the concentration of soluble organics in wastewater by lime coagulation increased polio-virus removal in studies by Gerba et al. (466, 467). In addition, the degree of virus adsorption from lime-treated wastewater exceeded that from filtered wastewater. These investigations also showed that poliovirus removal from wastewater effluents is greatly improved by maintaining a pH value in the range of 3.5 to 4.5. It was found that virus adsorbed at low pH could become desorbed by a rise in pH.

TERTIARY TREATMENT: ION EXCHANGE

INTRODUCTION

The process of selective ion exchange has long been utilized in the treatment of industrial process waters and in domestic water supply softening. Ion exchange resins (1142) are classified by the charge of the exchangeable ion. Thus, resins may be either catonic or anionic. General purpose resins will selectively exchange both cations and anions. The operational features of the ion exchange process are well developed and reliable. Such systems offer a reliable method of removing inorganic contaminants from the wastewater stream.

As can be seen by an examination of Table 40, very little information is currently available on removals of contaminants from municipal wastewater by use of ion exchange techniques. The process has not been economically feasible for treatment of municipal wastewater. Several research programs focusing on the application of ion exchange to municipal wastewater treatment are presently under way. The most promising future application appears to be for ammonia or nitrate removals.

WATER QUALITY PARAMETERS

Eliassen and Tchobanoglous (374, 375) conducted a review of the literature. They found that removals of phosphorus and nitrogen by tertiary wastewater treatment incorporating ion exchange can reach 90 percent. The actual removal efficiency was seen to depend upon the type of preceding treatment. Evans (397) investigated the removal of nitrate by ion exchange, concluding that the strong acid/weak base ion exchange process is well suited for this purpose. With the exception of these few studies of phosphorus and nitrate, most research performed to date has focused on ammonium removal, since specific exchange resins are not available for either the phosphorus or nitrate ions. However, some zeolite exchange resins do have unusual selectivity for the ammonium ion. This fact has encouraged research activity.

On the basis of both pilot and laboratory scale investigations, it appears that effluent ammonia concentrations of less than 1 mg/ℓ can be expected with ion exchange (730, 885). In a pilot plant study, Mercer et al. (898) used zeolite columns to test secondary effluent containing 10-19 mg/ℓ ammonia. Greater

TABLE 40 . LITERATURE REVIEWED PERTAINING TO ION EXCHANGE

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	4, 120, 165, 196, 277, 374, 454, 730, 885, 898, 1085, 1142, 1501
BOD	119, 120, 196, 277, 487, 885
COD	119, 120, 196, 277, 376, 885, 1547
Chlorides	376, 885
Nitrates	4, 120, 165, 374, 375, 397, 1142, 1501
Nitrites	165, 374, 1501
Phosphates	119, 120, 196, 277, 374, 375, 376, 379, 397, 463, 885
Suspended solids	4, 119, 120, 196
Total dissolved solids	196, 224, 277, 454, 1547
Total organic carbon	119, 120
Other (general)	531, 622
Elemental Contaminants	
Arsenic	520, 1062, 1244
Boron	1140
Cadmium	753, 802, 1062
Chromium	753, 802, 1062, 1157, 1169
Cobalt	753
Iron	753
Lead	753
Manganese	753
Mercury	753, 802, 912, 1062

TABLE 40 (continued)

Contaminant	Reference Number
Elemental Contaminants	
Nickel	753
Selenium	802
Other (general)	760, 1062
Synthetic/Organic Contaminants	515, 1153

than 99 percent removal of ammonia was achieved. Similarly, 99.7 percent of the ammonia in activated carbon effluent was removed by a zeolite in laboratory scale experiments by McKendrick et al. (885).

The Environmental Protection Agency (1085) reviewed pilot plant studies involving the use of clinoptilolite - a naturally occurring zeolite - for wastewater treatment. Ammonia removals ranged from 93 to 97 percent.

It should be noted that the ion exchange process using a zeolite such as clinoptilolite does not result in the production of a sludge containing the removed ammonia. Rather, the spent zeolite is regenerated with a lime slurry, which is subsequently air stripped, discharging ammonia to the atmosphere.

ELEMENTAL CONTAMINANTS

Ion exchange techniques have been principally applied for the removal of elemental contaminants from industrial waste streams (1062). Few studies have dealt with the application of ion exchange techniques to municipal wastewaters for elemental contaminant removal. Lindstedt et al. (802) investigated trace metal removal and concluded that a cation-anion exchange sequence was effective in reducing the concentrations of cadmium, chromium, and selenium in secondary effluent. Removal efficiencies are summarized in Table 41.

TABLE 41. TRACE METAL REMOVALS BY ION EXCHANGE (802)

Trace Metal	Percent Removal Cation Exchange	After Given Process Cation-Anion Exchange
Cadmium	99	99.9
Chromium	5	96
Selenium	1	99.7

Shen (1244) studied the removal of arsenic from drinking water by ion exchange, concluding that the process does not effect adequate removals. Using a variety of input levels, he was able to achieve arsenic reduction of only 21 percent.

BIOCIDAL CONTAMINANTS

Biocidal contaminant removal through ion exchange has also received little attention in the literature. In the only study located, Grover (515) stated that trifluralin, triallate, diallate, and nonionic herbicides were readily adsorbed on both cationic and anionic exchange resins, with somewhat more adsorption occurring on the cationic than on the anionic form.

TERTIARY TREATMENT: NITROGEN REMOVAL PROCESSES

INTRODUCTION

Interim primary drinking water standards established by the EPA set a nitrate limit of 10 mg/ℓ in the nitrogen form. Nitrogen concentrations in raw municipal wastewaters generally exceed this value, ranging from 15 to 50 mg/ℓ. Unless facilities are specifically designed to remove nitrogen, much of it will remain essentially unaffected, passing through the varying stages of treatment to ultimately enter the environment. Moreover, reuse of wastewater treatment plant effluents for direct groundwater recharge, indirect groundwater recharge through land application, or indirect reuse as a potable water supply is on the increase. Such reuse policies make effective nitrogen removal an important aspect of any wastewater treatment scheme.

In raw municipal wastewater, nitrogen is primarily found in the form of both soluble and particulate organic nitrogen and as ammonium ions. Conventional primary and secondary treatment transforms some of this organic nitrogen into ammonium ions. Part of the ammonium ion is oxidized to nitrate, and about 15 to 30 percent of the total nitrogen is removed.

Tertiary treatment processes designed to remove wastewater constituents other than nitrogen often remove some nitrogen compounds as well. However, removal is often restricted to particulate forms, and overall efficiency is generally low. Two tertiary processes particularly designed to remove nitrogen have been developed: nitrification-denitrification and ammonia stripping. Tertiary nitrification-denitrification usually involves two stages. Nitrification occurs in an initial stage, during which ammonium ions are oxidized to nitrite and nitrate ions by nitrifying bacteria. These nitrite and nitrate ions are in turn reduced to nitrogen gas which simply escapes from the system.

Ammonia stripping is effective only in removing ammonia nitrogen from municipal wastewater and has no effect on organic nitrogen, nitrite, or nitrate. Several ammonia stripping plants are in operation in the U.S. (Lake Tahoe, California, Orange County, California), but the process has been found to be expensive. A number of technical problems remain to be solved as well (885).

Nitrification and denitrification are biological reactions which occur naturally during several conventional treatment processes such as activated sludge treatments, aerobic lagooning, and anaerobic digestion. The activated sludge process, in particular, can be closely controlled to promote nitrogen removal. Such treatment processes are not principally designed to remove nitrogen, and both nitrification and denitrification occur only as secondary reactions. For ease of reference, however, all literature reviewed on the general topic of nitrogen removal has been tabulated on Table 42 , including tertiary processes specifically designed for this purpose.

WATER QUALITY PARAMETERS

There is general agreement that a system incorporating secondary biological treatment and tertiary nitrification-denitrification should achieve 80 to 95 percent total nitrogen removal at design flows (531,1001). The nitrification process alone removes only 5 to 10 percent of the total nitrogen entering the process, while oxidizing up to 98 percent of the ammonia nitrogen present to nitrate (458).

Average nitrogen data from systems incorporating nitrification-denitrification processes recorded by an EPA Technology Transfer Publication (458) are presented in Table 43 . Based on this report, the predicted effluent quality from a nitrogen-denitrification system will be 1.0 mg/ℓ organic nitrogen, 0.5 mg/ℓ ammonia nitrogen, 0.5 mg/ℓ nitrate nitrogen, and 2.0 mg/ℓ total nitrogen.

Nitrification is in itself an oxygen-demanding process and therefore reduces the total oxygen demand (TOD) in the wastewater effluent. Conventional biological or physico-chemical treatment obtaining up to 90 percent BOD reduction will only partially reduce the TOD of treated wastewater. For instance, such treatment will only reduce an influent TOD of 490 mg/ℓ to an effluent TOD of over 100 mg/ℓ. Nitrification will reduce the TOD of this effluent to less than 40 mg/ℓ (458).

Since the denitrification step involves the oxidation of carbonaceous material, a reduction in biochemical oxygen demand and total organic carbon can also be expected, in addition to the effective reduction of TOD.

Nitrogen removal by ammonia stripping was studied by O'Farrell et al. (1016), who reported a 90 percent removal of ammonia from a non-nitrified, lime-clarified secondary effluent at pH 11.5. During a warm weather study performed at Lake Tahoe, stripping produced a 95 percent removal of ammonia nitrogen at pH 11.5 and using 400 cu ft of air per gallon of wastewater (531). Any arbitrary percentage removal can be achieved with this type of system within available engineering capabilities, although higher removals mean higher costs.

TABLE 42. LITERATURE REVIEWED PERTAINING TO NITROGEN REMOVAL

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	4, 5, 8, 81, 121, 236, 374, 390, 458, 531, 643, 869, 885, 986, 1001, 1016, 1085, 1106, 1142, 1282, 1333, 1351, 1482, 1501, 1546
BOD	81, 458, 646
Chlorides	819
Nitrates	4, 5, 81, 374, 375, 390, 458, 489, 490, 642, 643, 646, 869, 986, 1001, 1085, 1106, 1142, 1282, 1350, 1351, 1501
Phosphates	374
Suspended solids	81, 646, 1350

TABLE 43. EFFLUENT NITROGEN CONCENTRATIONS IN TREATMENT SYSTEMS INCORPORATING NITRIFICATION - DENITRIFICATION (458)

	Average Effluent Nitrogen, mg/ℓ				
Type and Process Sequence	Organic-N	NH_4^+-N	NO_3^--N	NO_2^--N	Total N
Lime treatment of raw sewage, nitrification, denitrification	1.1	0.3	0.5	0.0	1.9
Primary treatment, high rate activated sludge, nitrification, denitrification, filtration	0.8	0.0	0.7	0.0	1.5
Primary treatment, roughing filters, nitrification, denitrification, filtration	0.8	0.9	0.6	0.0	2.3

DISINFECTION: CHLORINATION

INTRODUCTION

Until recently, chlorination was considered virtually an unmixed blessing as a cheap, effective method to destroy bacteria and viruses. It is now recognized, however, that chlorination of wastewater may create chlorinated compounds harmful to the environment and to human health. The extent of this potential hazard has not yet been determined; new and existing wastewater treatment plants continue to utilize chlorine for disinfection. The primary purpose of municipal wastewater chlorination is the destruction of pathogenic microorganisms. This is reflected in the literature reviewed, shown in Table 44.

WATER QUALITY PARAMETERS

Zaloum and Murphy (1553) concluded that chlorination of treated wastewater effluents does not reduce BOD, COD, and total organic carbon. Susag (1346), however, found BOD reductions by chlorination of up to 2 mg/ℓ of chlorine were added. These values are somewhat misleading, in that BOD reduction was due both to oxidation of the organic material and to the formation of chlorinated organics resistant to bacterial action.

When chlorine is added to a wastewater containing ammonia nitrogen, ammonia reacts with the hypochlorous acid formed to produce chloramines. Further addition of chlorine converts the chloramines to nitrogen gas. The reaction is influenced by pH, temperature, contact time, and initial chlorine-to-ammonia ratio. If sufficient chlorine is added, 95 to 99 percent of the ammonia will be converted to nitrogen gas with no significant formation of nitrous oxide. The quantity of chlorine required was found to be 10 parts by weight of chlorine to 1 part of ammonia nitrogen when treating raw sewage. This ratio decreased to 9:1 for secondary effluents, and 8:1 for lime-clarified and filtered secondary effluent (1346).

ELEMENTAL CONTAMINANTS

Little information is available on the minimal removal by chlorination of elemental contaminants. Andelman (27) studied the effects of chlorination on barium, copper, and nickel. The treatment effected a 34 percent reduction in barium, a 5 percent reduction in nickel, and had no effect upon copper. Kokoropoulos

TABLE 44. LITERATURE REVIEWED PERTAINING TO CHLORINATION

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	43, 76, 114, 120, 127, 727, 728, 1085, 1116, 1288, 1555
BOD	120, 321, 593, 1046, 1084, 1215, 1346, 1553
COD	120, 1084, 1091, 1346, 1553
Chlorides	684
Cyanides	65, 1046
Nitrates	120, 684, 1085, 1322
Nitrites	727, 728, 1085
Phosphates	120, 684
Suspended solids	120, 1084, 1215
Total dissolved solids	1215
Total organic carbon	120, 1553
Others (general)	385, 391, 531, 727, 728, 884, 1468
Elemental Contaminants	
Barium	27
Boron	1367
Copper	27
Iron	727, 1046
Manganese	727, 728, 1181
Mercury	1046, 1499
Nickel	27, 728

TABLE 44 (continued)

Contaminant	Reference Number
Synthetic/Organic Contaminants	6, 84, 158, 191, 482, 657, 971, 1130, 1250, 1287, 1332, 1450, 1555
Biological Contaminants	
Adeno virus	95
Bacteria	8, 79, 161, 210, 213, 283, 292, 321, 395, 615, 651, 830, 1304
Coliforms	161, 164, 292, 321, 395, 383, 395, 483, 612, 615, 727, 728, 937, 1084, 1304, 1312, 1389, 1393, 1458, 1489, 1490
Coxsackie virus (A & B)	95, 257, 380, 615, 890, 1206
ECHO virus	380, 615, 1260
Escherichia coli	141, 375, 383, 428, 615, 1206, 1414, 1430
Fecal streptococci	292, 395, 612, 1084
Hepatitis virus	615
Mycobacterium	428, 615
Parasitic worms	161, 615, 1188, 1304
Polio virus	161, 223, 380, 615, 817, 820, 1206, 1260, 1293, 1425
Protozoa	161, 383, 428, 1335
Salmonella	161, 292, 395, 428, 1083
Shigella	161
Vibrio cholerae	161
Virus	79, 92, 95, 96, 100, 101, 127, 161, 210, 213, 280, 281, 283, 352, 382, 383, 395, 454, 492, 615, 651, 739, 745, 816, 817, 900, 970, 1009, 1083, 1095, 1243, 1259, 1260, 1304, 1366, 1434
Other (general)	161, 588

(727) reported hypochlorous acid reacted with soluble iron (II) and manganese (II) to form precipitates.

SYNTHETIC/ORGANIC CONTAMINANTS

No research was found to address removal or destruction effects of chlorination on any of the synthetic/organic contaminants. However, considerable interest has recently developed concerning the production of chlorine-containing organic compounds by chlorination. The reactions of chlorine with organic compounds in water are diverse, including oxidation, substitution, addition, and free radical reactions. Chlorination may produce several different chlorinated products from a single organic pollutant molecule. Some of these compounds have been identified as toxic to aquatic life by Snoeyink (1287), Brungs (158), and others.

Jolley (657) evaluated chlorine-containing organic constituents in chlorinated effluents and found that stable chlorine-containing compounds were present after effluents had been chlorinated to a 1 to 2 mg/ℓ chlorine residual. These compounds are identified in Table 45.

TABLE 45. IDENTIFICATION OF CHLORINE CONTAINING CONSTITUENTS IN CHLORINATED EFFLUENTS (657)

5 - Chlorouracil
8 - Chlorocaffeine
8 - Chloroxanthine
5 - Chlorosalicylic acid
2 - Chlorophenol
4 - Chlorobenzoic acid
3 - Chlorobenzoic acid
3 - Chloro-4-hydroxybenzoic acid
4 - Chloro-3-methylphenol
5 - Chlorouridine
6 - Chloroguanine
2 - Chlorobenzoic acid
4 - Chloromandelic acid
4 - Chlorophenylacetic acid
4 - Chlorophenol
3 - Chlororesorcinol

A similar project was conducted by Glaze and Henderson (482). The chlorinated organics identified in this study are listed in Table 46.

Shimizu et al. (1250) stated that halogenated nucleic acid bases are incorporated into the nucleic acid. Also, the incorporation of 5-deoxybromouridine in DNA and 5-fluorouracil into RNA are known to cause mutations. No work has been completed to determine how nucleic acids react with chlorine or the resulting mutations.

TABLE 46. CHLORINATED ORGANICS IN WASTEWATER EFFLUENT (482)

Chloroform	Dibromochloromethane
Dichlorobutane 3-chl	3-chloro-2-methylbut-1-ene
Chlorocyclohexane (-18)	Chloroalkyl acetate
O-dichlorobenzene	Tetrachloroacetone
P-dichlorobenzene	Chloroethylbenzene
Pentachloroacetone	Hexachloroacetone
Trichlorobenzine	Dichloroethyl benzene
Chlorocumene	
N-methyl-trichloroaniline	Dichlorotoluene
Trichlorophenol	
Chloro-a-methyl benzyl alcohol	
Dichloromethoxytoluene	
Trichloromethylstyrene	Trichloroethyl benzene
Dichloro-a-methyl benzyl alcohol	
Dichloro-bis (ethoxy) benzene	
Dichloro-a-methyl benzyl alcohol	Trichloro-N-methylanisole
Trichloro-a-methyl benzyl alcohol	Tetrachlorophenol
Trichloro-a-methyl benzyl alcohol	Trichlorocumeme
Tetrachloroethylstyrene	Trichlorodimethoxybenzene
Tetrachloromethoxytoluene	
Dichloroaniline derivative	
Dichloroaniline derivative	Dichloroacetate derivative
Trichlorophthalate derivative	
Tetrachlorophthalate derivative	

BIOLOGICAL CONTAMINANTS

The effectiveness of chlorination as a disinfection process has long been recognized. All researchers are in agreement that the effectiveness of disinfection by chlorine is influenced by time and chlorine concentration and also by: (1) whether the chlorine residual is free or combined; (2) effectiveness of mixing; (3) whether or not particulates are present; (4) pH; (5) temperature; and (6) the concentration, condition, and nature of the organisms. Keeping these limitations in mind, an idea of the relative resistances of organisms to disinfection by chlorine can be seen in Table 47.

TABLE 47. EFFECT OF CHLORINATION ON VARIOUS ORGANISMS (615)

Group	Organism	Chlorine Residual (mg/ℓ)	Time min.	Efficiency
Virus	Infectious hepatitis	1	30	Survived
		15	30	Inactivated
	Coxsackie	5	2.5	Survived
	Coxsackie	1.0	3	99.6% Inactivated
	Echo	1.95	6.5	Survived
	Poliovirus I	0.53	14	Survived
	Coliphage B	0.03	10	20% Survival
	Theiler Phage	0.03	10	Inactivated
Bacteria	*M. tuberculosis*	1-5	120	99% Kill
		2	30	99% Kill
		1	30	Destroyed
	E. coli	0.14	3	99.9% Kill
	Coliforms	0.03	10	52% Kill
		1-1.2	15	99% Kill
	Total Count	some	15	98-99% Kill
Nematodes	Diplogaster	2.5-3	120	Survived and Mobile
	Cheilobus	15-45	1	
Others	*S. mansoni* (ova and miracidia)	0.2-0.6	30	Killed
	S. japonicum (ova and miracidia)	0.2-0.6	30	Killed

Eliassen and Tchobanoglous (375) found that 2 to 6 mg/ℓ of chlorine applied for 20 min would effect a 99.99 percent kill of the total coliforms, fecal coliforms, and fecal streptococci present in wastewater influent.

The effect of chlorination on entamoebic cysts was investigated by Stringer and Kruse (1335). It was concluded that the hypochlorous acid (HOCl) form of free available chlorine was the most rapidly acting cysticide compared with other halogen species. Entamoeba histolytica cysts and tapeworm eggs withstand the chlorination treatment generally applied to waste treatment effluent (161). Rudolfs et al. (1188) reported the development of active embryos from a majority of ascaris eggs in contact with 5,000 ppm chlorine solutions for 30 min.

Davis and Keen (292) conducted a research project on the ability of municipal wastewater bacteria to survive chlorination and reestablish populations after discharge. Fecal coliform, fecal streptococci, and total coliform enumeration was performed with further differentiation into lactose nonfermenters within and outside the family Enterobacteriaceae. It was determined that the vast majority of the wastewater bacterial species that reestablished their populations within 21 days following chlorination were lactose nonfermenters not included in the Enterobacteriaceae. Many of these bacteria could be pathogenic under the appropriate conditions and may constitute a threat to public health in receiving waters designated for contact recreation.

Virus inactivation is one of the more difficult tasks of chlorination, but as in any disinfection process, required kills can be achieved by lengthening the time or increasing the concentration (615). A study of the inactivation of viruses in wastewaters by chlorination was performed by Lothrup and Sproul (817). It was ascertained that:

- High-level inactivation of viruses can be obtained in treated and untreated domestic wastewaters. Present chlorination practices (1 mg/ℓ of residual), however, are inadequate for a high level of virus inactivation.

- A combined chlorine residual of 28 mg/ℓ was required to produce a 99.99 percent inactivation of the T2 bacteriophage in settled raw wastewater after a 30-min contact time.

- A combined chlorine residual of 40 mg/ℓ was required to provide a 99.99 percent destruction of the Type I polio virus in settled wastewater after a 30-min contact time.

- Free chlorine residuals of 0.2 to 0.4 mg/ℓ , after 30 min, produced a complete inactivation of the polio virus and T2 phage in the secondary effluent.

- In experimental runs with synthesized storm-water overflow samples, a 100 percent inactivation of the Type I polio virus was obtained by providing a free chlorine residual.

- The T2 bacteriophage was much less sensitive to combined chlorine residuals than is the coliform organism and somewhat more sensitive than the polio virus to combined chlorine residuals.

A similar research project investigating the inactivation of enteroviruses by chlorination was conducted by Shuval et al. (1260). The following conclusions were obtained:

- The strain of ECHO virus used was sensitive to the combined chlorine in the sewage, with reductions of 99 percent in 30 min and 99.93 percent in 6 hr using 3.6 mg/ℓ of chlorine. No virus was recovered in the sample tested after 4 hr contact with 7 mg/ℓ of chlorine dosage or after 2.5 hr with 11 mg/ℓ of applied chlorine. Inactivation was shown to be a function of time and chlorine concentration under the test conditions.

- The strain of polio virus Type I used was much less sensitive to the combined chlorine, showing only 50 and 90 percent reductions in 6 hr using 4 and 11 mg/ℓ chlorine dosage respectively.

- Coliform reductions obtained followed known patterns, with a standard of less than 100 coliforms/100 mℓ being obtained in 80 percent of the samples after 2 hr of contact with a chlorine dose of about 8 mg/ℓ.

- Although the inactivation of the ECHO virus strain was comparable to that obtained with coliform organisms, this study indicates that the coliform index in chlorinated sewage may not give a true picture of the degree of inactivation obtained with the more resistant strains of enterovirus such as polio virus Type 1.

The minimum concentration of chlorine required for complete inactivation of the Sabin oral poliovaccine Type I virus strain was examined by Varma et al. (1425). Various exposure periods with pH 5.2 at 20°F were studied: a concentration of 22 mg/ℓ for 5 min of exposure time, 19 mg/ℓ for 15 min, 19 mg/ℓ for 30 min, 17 mg/ℓ for 45 min, and 14 mg/ℓ for 60 min. Nonetheless, on the basis of the literature surveyed, it is evident that chlorination per se does not provide conclusive proof of disinfection.

Boardman and Sproul (127) described the protection afforded viruses associated in particulate matter. Surface adsorption gave no viral protection. When viruses are embedded within particles, the disinfecting molecules must diffuse through the particle matrix before reaching the virus and initiating any inactivation. Chemical diffusion is a slow process; as a consequence, virtually all the embedded viruses are prtected from disinfection.

DISINFECTION: OZONATION

INTRODUCTION

Ozone, an allotropic form of oxygen, is a powerful oxidizing agent for the disinfection of wastewater. Ozone is used in over 100 municipalities in Europe for disinfection of drinking water. Certain chemical features make ozone treatment a particularly attractive method of water purification:

- It is a powerful oxidant which reacts rapidly with most organic compounds and microorganisms in wastewater.
- It does not impart taste and odor to potable water.
- It is produced from oxygen in air by means of electric energy.

On the negative side, the cost of ozonation is not presently competitive with chlorine disinfection. Moreover, long-term residual disinfection capabilities are lacking, and the instability of ozone generally necessitates its generation on site (1429).

The principal ozone decomposition products in aqueous solution are molecular oxygen and the highly reactive free radicals OH, HO_2, HO_3+. Very little is known about the significance of the free radical intermediates on the germicidal properties of ozone solutions. The same free radicals are produced by irradiation of water, and it has been reported that HO_2 and OH. radicals contribute significantly to the killing of bacteria by this process.

As seen in Table 48, a considerable amount of information is available on the destruction of various pathogens by ozonation, however little information was found on the effect of ozone upon other contaminants.

WATER QUALITY PARAMETERS

Because of its strong oxidizing character, ozone is very reactive toward the organic compounds which make up the BOD, COD, and total organic carbon. Under ideal conditions the reactions would result in almost complete oxidation and only carbon dioxide as a reaction product. In practice, ozonation results

TABLE 48. LITERATURE REVIEWED PERTAINING TO OZONATION

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	8, 470, 651, 709, 1166
BOD	320, 321, 709, 868, 970, 985
COD	236, 320, 470, 707, 985, 1231
Nitrates	470, 651, 985, 1166
Nitrites	470, 651, 709, 1166
Phosphates	985
Suspended solids	470
Total organic carbon	985, 1262
Others (general)	394
Elemental Contaminants	437, 438
Adeno virus	95, 213
Bacteria	8, 72, 213, 320, 321, 469, 542, 707, 709, 1178
Clostridium botulinium	1429
ECHO virus	1429
Escherichia coli	72, 95, 686, 687, 707, 1429
Fecal streptococci	72, 707
Parasitic worms	897, 1429
Polio virus	213, 223, 686, 687, 835
Salmonella	707
Virus	95, 96, 97, 213, 280, 320, 492, 816, 818, 1069, 1070
Other (general)	686

in only partial oxidation and produces simpler organic molecules. Both Ghan and Nebel reported COD removals of less than 40 percent. Morris found that the apparent BOD of a wastewater can increase after ozonation as a result of refractory organic molecules being oxidized to simpler, biodegradable compounds. If the ozone is applied after other treatment processes (as is normal), the increase in organic nutrient molecules can lead to the growth in the distribution system of algae, slime bacteria, and the possible regrowth of any pathogens not destroyed during treatment.

Ozone is effective at decreasing concentrations of organic suspended solids and organic nitrogen through oxidation. Ozonation can assist in suspended solids removal through froth flotation mechanisms induced through the process. Ozone will also oxidize nitrites to nitrates, but will not react with ammonia (651). There is little evidence to date that ozonation will produce any toxic or carcinogenic oxidation by-products as will chlorination.

ELEMENTAL CONTAMINANTS

Furgason and Day (438) studied the feasibility of ozonation for iron and manganese removal from raw water with relatively high input concentrations. The study demonstrated that ozone effectively oxidized the iron and manganese to an insoluble form which could be filtered from the water.

Complete oxidation of the minerals required a reaction time of 30 sec. Filtration studies indicated a relatively fine medium was required to remove the oxide precipitate.

BIOLOGICAL CONTAMINANTS

The use of ozone as a wastewater disinfectant was reviewed by Venosa (1429). It was concluded that with 0.1 mg/ℓ of active chlorine, 4 hr would be required to kill 6 x 10^4 E. coli cells in water, whereas with 0.1 mg/ℓ of ozone only 5 sec would be necessary. When the temperature was raised from 22°C to 37°C, the ozone inactivation time decreased from 5 sec to 0.5 sec. These investigations revealed that the contact time with ozone necessary for 99 percent destruction of E. coli was only one-seventh that observed with the same concentration of hypochlorous acid. The death rate for spores of Bacillus species was about 300 times greater with ozone than with chlorine.

In the same study, Venosa also described bacteriological studies performed on secondary effluent from an extended aeration pilot plant in the Metropolitan Sewer District of Louisville, Kentucky. Using an average applied ozone dosage of 15.2 mg/ℓ for an average contact time of 22 min, fecal coliform reductions of greater than 99 percent were achieved, resulting in a mean fecal coliform concentration of 103 cells/100 mℓ, a mean total coliform concentration of 500 cells/100 mℓ, and a mean fecal streptococci concentration of 8 cells/100 mℓ in the final effluent. Laboratory results with raw sewage indicated that ozone could be successfully used to sterilize sewage containing *Bacillus anthracis*, influenza virus, and *B. subtilis* morph. *globibii*, and to inactivate toxins of *Clostridium botulinum*. Ozone consumption was 100 to 200 mg/ℓ for 30 min. Finally, Venosa found ozone to be many times more effective than chlorine in inactivating poliomyelitis virus. Identical dilutions of the same strain and pool of virus, when exposed to 0.5 to 1.0 mg/ℓ of chlorine and 0.05 to 0.45 mg/ℓ of ozone, were devitalized within 1.5 to 2 hr by chlorine, while only 2 min of exposure were required with ozone.

Majumdar et al. (835) also studied the inactivation of polio virus by ozonation and concluded that the inactivation is not complete. Results are summarized in Table 49.

TABLE 49. SURVIVAL OF POLIO VIRUS IN OZONATION CONTINUOUS FLOW STUDIES (835)

Type of Wastewater	Ozone Concentration (mg/ℓ)	Residence Time (min)	Average Survival (percent)
Primary wastewater	0.84	8.0	1.820
	1.47	2.0	0.016
	4.44	1.0	0.006
Secondary wastewater	0.79	8.0	2.055
	1.77	2.0	0.013
	5.05	1.0	0.006

Pavoni and Tittlebaum (1069) recently studied ozone disinfection of viruses in the Fort Southworth Pilot Plant of the Metropolitan Sewer District in Louisville. Using F2 bacteriophage as the model virus, they demonstrated virtually 100 percent inactivation efficiency in the secondary effluent after a contact time of 5 min at a ozone dosage of approximately 15 mg/ℓ and a residual of 0.015 mg/ℓ. Of particular interest was the observation that the rate of inactivation was greater for F2 bacteriophages than bacteria. In addition, the following conclusions were reached:

1. F2 virus concentrations were shown to be unaffected by the flow or mixing of the ozone reactor.

2. F2 virus was inactivated with virtually 100 percent efficiency after a contact time of 5 min at a total ozone dosage of approximately 15 mg/ℓ and a residual of 0.015 mg/ℓ .

3. E. coli bacteria and F2 virus were inactivated with virtually 100 percent efficiency after a contact time of less than 15 sec in the absence of ozone-demanding material.

4. An extremely small number of viral particles was observed in effluent studies.

5. Oxidation by ozone appears to be the mechanism of kill for bacterial cells and viral particles. Ozone is theorized to act as a general oxidant causing cell lysis and the release of soluble COD.

Mercado-Burgos et al. (897) examined the effect of ozone on Schistosoma ova, concluding that the process was ineffective.

SLUDGE TREATMENT AND DISPOSAL SYSTEMS

THICKENING AND DEWATERING

INTRODUCTION

Sludge is the most notable by-product of the wastewater treatment plant. Disposal of sludge materials is a problem comparable in magnitude and importance to that of wastewater treatment. Because of the high water content of sludge, any reduction in volume through dewatering or thickening, of these by-products marks a significant step towards an improved solution of the disposal problem. Processes available for sludge volume reduction include:

- Gravity sludge thickeners
- Air flotation sludge thickeners
- Centrifugal sludge thickeners
- Vacuum filters
- Drying lagoons
- Pressure filtration

Sludge dewatering by land methods (e.g., lagooning) has been the accepted practice for many years. However, attention has recently focused on the use of other techniques available for dewatering. Increased land costs together with stringent land, air, and water pollution standards have made sludge dewatering an economically attractive process since it significantly reduces the volume of sludge requiring disposal.

Literature concerning sludge thickening and dewatering processes, as reviewed in Table 50 , is rarely pertinent to discussions of adverse health effects of sludge-contained contaminants. In general, this literature deals mainly with engineering design and operation parameters and the relative efficiency of dewatering processes, with only slight coverage of partitioning, degradation, or concentration of either general, elemental, or biological contaminants.

Coagulants are used for improved dewatering characteristics and have been indicated as possible factors providing viral inactivation or removal (246). Kampelmacher and Jansen (670) reported reduction of bacteria in sludge through vacuum filtration and chemical conditioning by ferric chloride,

TABLE 50. LITERATURE REVIEWED PERTAINING TO SLUDGE THICKENING AND DEWATERING

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	925, 1080
BOD	16, 201, 334, 1435
COD	16, 334
Chlorides	1080
Nitrates	334
Phosphates	546, 1080
Suspended solids	16, 201, 229, 438, 518, 546, 864, 1008, 1015, 1054, 1058, 1080, 1123, 1308, 1368, 1369, 1435
Others (general)	16, 201, 334, 401, 518, 531, 546, 551, 552, 622, 638, 639, 764, 1008, 1015, 1053, 1054, 1055, 1058, 1123, 1308, 1369, 1397
Elemental Contaminants	
Aluminum	246, 1080
Boron	1080
Cadmium	229, 1080
Chromium	229, 1080
Copper	229, 635, 1080
Iron	229, 1080
Lead	1080
Manganese	229, 1080
Mercury	635
Nickel	635, 1080

TABLE 50 (continued)

Contaminant	Reference Number
Zinc	229, 635, 1080
Other (general)	229
Biological Contaminants	
Bacteria	670
Coliforms	401, 568
Coxsackie virus (A & B)	1053
Escherichia coli	568
Salmonella	244, 670
Virus	246

ferrous sulfate, and lime. Two separate analyses of Dutch treatment plants showed a marked reduction (from two to four orders of magnitude) of aerobic bacteria and enterobacteriaceae.

Peterson et al. (1080) studied the chemical properties of waste activated sludge after vacuum filtration; his results, presented in Table 51, provide a representative picture of the constituents found in thickened and dewatered sludge.

TABLE 51 . CHEMICAL PROPERTIES OF VACUUM FILTERED WASTE ACTIVATED SLUDGE* (1080)

Analyses	% of Dry Wt Basis
Total Nitrogen	6.37
Ammonia Nitrogen	Trace
Phosphorus	2.49
Zinc	--
Boron	0.002-0.04
Iron	5.32
Manganese	0.012
Aluminum	--
Cadmium	0.028
Chloride	--
Chromium	0.362
Copper	0.11
Nickel	0.034
Lead	0.141

* Waste activated sludge; ferric chloride addition; vacuum filtered; heat dried

ANAEROBIC DIGESTION

INTRODUCTION

Sludge digestion converts bulky, odorous, raw wastewater sludge to a relatively stable material that can be readily dewatered and disposed without excess obnoxious odors. Bacterial digestion is a two-stage process effecting the decomposition of organic and/or inorganic material in the absence of free oxygen. The first stage is accomplished by facultative bacteria, which break down large organic compounds and convert them to simpler organic acids. Acid-splitting, methane-forming, anaerobic bacteria complete the second step, converting the organic acids to methane and carbon dioxide. These strict anaerobes have a relatively slow growth rate and are highly sensitive to environmental conditions of temperature, pH, and anaerobiosis.

Health-related problems traceable to anaerobic digestion would be dependent upon the final sludge solids disposal scheme, since the dewatering liquor circulates in a closed loop system within the treatment plant. Pertinent literature reviewed concerning this subject is indicated in Table 52.

WATER QUALITY PARAMETERS

Sekikawa et al. (1230) investigated the solubilization of certain water quality parameters under anaerobic conditions. A period of marked increase in dissolved phosphate concentration and BOD value was noted during the initial stage of digestion. In the ensuing stage, these values decrease. The maximum amount of phosphorus released equaled approximately 50 percent of the total phosphorus contained in the sludge. These temporary increases were attributed to autolysis, or microbial destruction, of sludge organisms. Table 53 shows typical anaerobic digestor content characteristics during the study.

ELEMENTAL CONTAMINANTS

Anaerobically digested sludge characteristically contains high concentrations of elemental contaminants. Sludge volume is reduced during digestion, with a corresponding increase in metal concentrations. The extent to which these metals inhibit the digestion process, however, has in the past received more attention than any ultimate effect on human health.

TABLE 52 . LITERATURE REVIEWED PERTAINING TO ANAEROBIC DIGESTION

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	580, 584, 690, 864, 1080, 1190
BOD	205, 1080, 1230
COD	1080
Chlorides	1080
Nitrates	580, 584, 1195
Nitrites	112, 584
Phosphates	513, 580, 584, 623, 690, 736, 838, 1035, 1036, 1080, 1195, 1228, 1230, 1267
Suspended solids	497, 838, 1035, 1054, 1055, 1080, 1128, 1230
Others (general)	531, 551, 552, 622, 690, 1035, 1053, 1123, 1230
Elemental Contaminants	
Aluminum	513, 690, 1035, 1036, 1080, 1228
Barium	690
Boron	690, 1180, 1195
Cadmium	228, 580, 584, 690, 853, 975, 1063, 1080, 1195, 1400
Chromium	71, 228, 580, 584, 623, 690, 853, 1080, 1195, 1400
Cobalt	1195, 1400
Copper	71, 228, 584, 623, 690, 761, 853, 975, 1063, 1080, 1195, 1400, 1411
Iron	112, 228, 580, 690, 761, 975, 1080, 1228, 1267

TABLE 52 (continued)

Contaminant	Reference Number
Lead	228, 580, 584, 690, 853, 1063, 1080, 1195, 1400
Manganese	112, 228, 580, 584, 623, 690, 1080, 1195, 1400
Mercury	228, 690, 803, 1195
Nickel	71, 228, 584, 623, 690, 761, 1063, 1080, 1195, 1400
Zinc	71, 112, 228, 401, 584, 623, 690, 761, 853, 975, 1063, 1080, 1195, 1228, 1400
Other (general)	1228
Biocidal Contaminants	
DDT	14, 15
Synthetic/Organic Contaminants	714
Biological Contaminants	
Bacteria	161, 580, 935, 1123
Coliforms	568, 580, 582, 935
Coxsackie virus (A & B)	95, 112, 161, 428, 1053
ECHO virus	112
Escherichia coli	568, 1080, 1123
Mycobacterium	428
Parasitic worms	161, 428, 791
Polio virus	112, 161, 428
Protozoa	428, 1080
Salmonella	95, 161, 428, 717, 1080, 1123
Shigella	161

TABLE 52 (continued)

Contaminant	Reference Number
Vibrio cholerae	161
Virus	95, 96, 112, 161, 428, 580, 908, 1209, 1476
Other (general)	161

TABLE 53 . SLUDGE CHARACTERISTICS (1230)

Parameter	Concentration
Sludge volume index (%)	25.0
Total suspended solids (mg/ℓ)	2,784
Volatile suspended solids (mg/ℓ)	1,920
Ash (mg/ℓ)	864
Total nitrogen (mg/ℓ)	282
Total phosphorus (mg/ℓ)	60
as Phosphate	183.5
BOD (mg/ℓ)	1,465

Table 54 presents the concentration range of various constituents including elemental contaminants found in anaerobically digested liquid sludge from 35 Wisconsin municipalities, recorded by Keeney et al. (690). Table 55 shows the average metal concentrations reported by Salotto et al. (1195). The authors thought the geometric mean would be the best measure of central tendency. The spread associated with the geometric mean was greater for certain metals such as cadmium, chromium, and manganese, indicating that these metals -- not ordinarily found in domestic wastewater -- may occasionally be introduced (probably from industrial sources) in relatively high concentrations.

A laboratory investigation of mercury distribution in an anaerobic digester was reported by Lingle and Hermann (803). Data indicate that during experimentation, more than 96 percent of the mercury from phenyl mercuric chloride and mercuric chloride remained in the sludge solids. It is suggested that the biological methylation of mercury may have been inhibited by the presence of sulfides in the digester sludge. The

TABLE 54 . CONSTITUENT CONCENTRATIONS IN ANAEROBICALLY DIGESTED SLUDGE (690)

Constituent	Concentration Range*
Total nitrogen (moist)	3.4-9.5
Total nitrogen (dried)	2.4-3.1
Ammonia nitrogen (moist)	0.8-4.1
Ammonia nitrogen (dried)	0.02-0.26
Organic Carbon	25.7-38.5
Phosphorus	2.7-6.1
Aluminum	0.36-1.2
Iron	0.8-7.8
Cadmium/Zinc	0.15-33
Zinc	490-12,220
Copper	140-10,000
Nickel	15-1,700
Cadmium	5-400
Lead	40-4,600
Chromium	50-32,000
Silver	0.6-31
Boron	150-750
Manganese	180-1,130
Barium	530-1,340
Strontium	52-7,810

* Range for the first 9 constituents is given in percent of solids and in mg/kg for the last 11 constituents.

TABLE 55. AVERAGE CONCENTRATIONS OF METALS IN DIGESTED SLUDGE (1195)*

	Arithmetic		Geometrics		Median
Metal	Mean	Std. Dev. (+ and -)	Mean	Std. Dev. (÷ and x)	5% Value
Silver	250	230	190	1.99	100
Boron	430	310	380	1.58	350
Cadmium	75	104	43	2.47	31
Calcium	36,500	23,800	31,100	1.77	30,000
Chromium	1,860	1,920	1,050	3.22	1,100
Cobalt	350	220	290	1.88	<100
Copper	1,590	1,670	1,270	1.95	1,230
Mercury	10	18	6.5	2.34	6.6
Manganese	1,300	2,290	475	3.67	380
Nickel	680	620	530	1.88	410
Lead	2,750	2,350	2,210	1.82	830
Strontium	520	670	290	2.70	175
Zinc	4,210	3,800	2,900	2.40	1,780

* (All figures mg/kg dry sludge basis)

sulfide content of the sludge (228 mg/ℓ) may have precipitated the mercury, thus making it unavailable for biological methylation.

The use of sulfide as a detoxicant for other heavy metals has also been suggested, with the metal ions being precipitated out as metallic sulfides (552).

BIOLOGICAL CONTAMINANTS

The anaerobic digestion process effectively degrades proteins and other organic materials. Many studies have been made of the destruction or inactivation of viruses and bacteria. Hinesly et al. (580) reported the incidence of fecal coliforms to be approximately 10^5 cells/mℓ of digested sludge. The number of fecal coliforms in liquid digested sludge gradually decreases with increased digester retention time, as shown in Table 56.

TABLE 56. NUMBER OF FECAL COLIFORMS PER mℓ OF LIQUID DIGESTED SLUDGE AS A FUNCTION OF TIME (580)

Sludge samples	Days 0	19	32
Total sludge	4×10^4	7×10^3	2×10^2
Sludge supernatant	3×10^3	2×10^1	0

A study of the destruction of various species of bacteria by anaerobic digestion was performed for the EPA (1123). Results of the study (presented in Table 57) indicate that certain organisms were not entirely eliminated by digestion. This was corroborated in a review by Foster and Engelbrecht (428). The authors reported that destruction of _S. typhosa_ was 83 to 99.7 percent after 12 hr in batch digestion studies. A 31 percent survival of mycobacterium species was noted after digestion of more than 35 days. Tubercle bacilli were able to survive from 6.5 months up to 2 years after digestion, even though 90 percent of the organisms were destroyed during treatment.

Viral inactivation by anaerobic digestion is accomplished far more rapidly than bacteria destruction. The mechanism of infectivity loss or inactivation has not been determined, although the linearity of inactivation curves suggests that a single mechanism predominates (112).

Anaerobically digested sludge seeded with a swine enterovirus (EPCO-1) was fed to germ-free piglets in experiments performed by Meyer et al. (908). Indications of piglet infection were found after four but not five days of digestion. Based on the data provided, an inactivation rate of 91 percent per day can be estimated.

The inactivation of polio virus I, coxsackie virus A-9, coxsackie virus B-4, and ECHO virus II was studied by Bertucci et al. (112). According to this study, viral inactivation appeared to follow a first order reaction pattern. Significant differences were noted among the inactivation rates of the virus tested.

TABLE 57. BACTERIAL DESTRUCTION BY ANAEROBIC DIGESTION (1123)

Bacteria	Digestion Period (days)	Destruction %	Remarks
Endamoeba hystolytica	12	<100	Greatly reduced populations at 68°F
Salmonella typhosa	20	92	85% reduction in 6 days detention
Tubercle bacilli	35	85	Digestion cannot be relied upon for complete destruction
Escherichia coli	49	<100	Greatly reduced populations at 99°F, about the same reduction in 14 days at 72°F

AEROBIC DIGESTION

INTRODUCTION

Aerobic digestion is another method used to stabilize the organic fraction of wastewater treatment plant sludge. Available literature is listed on Table 58 . To date, this process has only been employed at small wastewater treatment plants or where polishing of excess waste-activated sludge is necessary. Widespread application of aerobic digestion is limited by the long solids retention time and consequential high oxygenation costs required by the process.

Aerobic digestion is a biological process during which oxidation is completed in two phases: (1) direct oxidation of any biodegradable matter by microorganisms; and (2) oxidation of microbial cellular material by endogenous respiration. After input waste organics are completely oxidized, the sludge mass is further reduced by endogenous metabolic activity.

Advantages of aerobic digestion include: (1) a volatile solids reduction comparable to that obtained by anaerobic digestion; (2) a humus-like digested sludge that has no disagreeable odor; (3) a relatively low BOD concentration in the supernatant; (4) a sludge that is easily dewatered; and (5) a relatively problem-free operation (531, 552, 622). However, the process has certain disadvantages: power costs to supply the required oxygen are high, and pH values as low as 4.5 may result due to the absence of methane gas production. Such extreme pH values inhibit nitrate formation and result in the release of large amounts of soluble phosphorus (552).

TABLE 58. LITERATURE REVIEWED PERTAINING TO AEROBIC DIGESTION

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	327
BOD	531, 552, 622, 1053
COD	456
Nitrates	327, 1281
Phosphates	552, 1230
Suspended solids	456, 531, 552, 622, 1053
Other (general)	531, 551, 552, 691, 1053, 1123
Elemental Contaminants	
Chromium	71
Copper	71
Nickel	71
Zinc	71
Biological Contaminants	
Bacteria	1281
Coliforms	568
Escherichia coli	568
Virus	1281

THERMAL PROCESSES

INTRODUCTION

Thermal processes, whether conventional heat treatment or incineration, use increased temperature to achieve either pasteurization or oxidation of organic material. Literature concerning these processes is shown in Table 59. Reports on heat treatment mainly cover the inactivation of biological contaminants; those on incineration more frequently address the subject of elemental contaminants.

Heat treatment is a well known method of destroying pathogenic organisms (1123). This wet air oxidation process is based on the principle that any substance capable of burning can be oxidized in the presence of liquid water at temperatures between 250°F and 700°F. The process can operate on difficult-to-dewater waste liquors and sludges with extremely low solids concentration. In general, given the proper temperature, pressure, reaction time, and sufficient compressed air or oxygen, any degree of oxidation desired can be accomplished. Flameless oxidation of organics can be achieved at relatively low temperatures of 300°F to 400°F with this technique, compared with the 1,500°F to 2,700°F necessary for conventional incineration. Air pollution is minimized since oxidation takes place in water at low temperatures, and no fly ash, dust, sulfur dioxide, or nitrogen oxide by-products are formed (1123).

Incineration has been a popular sludge treatment and disposal method for over 20 yr. Proper treatment of sludge prior to its introduction into the incinerator unit is the key to successful incineration. Pretreatment may include sludge thickening, a macerating or disintegrating system, and/or a dewatering device (1123).

The air emission products of sludge combustion are essentially steam, carbon dioxide, small amounts of particulate emissions, and oxide of sulfur (1017). Such by-products could pose problems for human health and environmental quality. Table 60 summarizes particulate, NO_x, sulfur dioxide, and visible emissions from four incineration sites. As shown, NO_x emissions are most significant.

TABLE 59. LITERATURE REVIEWED PERTAINING TO THERMAL PROCESSES

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	1236
BOD	864, 1435
COD	222
Nitrates	528, 788, 1281
Phosphates	507, 1354
Suspended solids	126, 1435
Other (general)	7, 88, 393, 531, 551, 552, 642, 864, 1017, 1123, 1354
Elemental Contaminants	
Aluminum	507
Boron	75
Cadmium	662, 926, 1017
Chromium	436, 746, 1017
Copper	401, 746, 1017
Iron	507, 1017
Lead	662, 746, 1017
Manganese	746
Mercury	401, 926, 1017, 1077
Nickel	401, 1040
Zinc	401, 436, 1040
Biocidal Contaminants	
Chlorinated hydrocarbons	401

TABLE 59 (continued)

Contaminant	Reference Number
DDT	401
Dieldrin	401
Synthetic/Organic Contaminants	567
Biological Contaminants	
Bacteria	1325
Coliforms	401
Escherichia coli	1123, 1325
Mycobacterium	1123, 1325
Parasitic worms	791, 1325
Protozoa	1325
Salmonella	1123, 1325
Virus	1123, 1281, 1325
Other (general)	1123, 1281, 1325

TABLE 60. EXHAUST EMISSIONS FROM SLUDGE INCINERATION (1017)

Emissions	Minimum	Maximum
% Carbon Dioxide - Vol. % (dry basis)	2.2	10.2
% Excess Air (test point)	366.0	51.0
Sulfur Dioxide (ppm)	1.97	14.26
NO_x (ppm)	11.47	271.63
Hydrochloric acid (ppm)	.621	11.9
Particulate - filter (GR/SCFD)	.0127	.0766
Particulate - total (GR/SCFD)	.0170	.0859
Sludge feed to furnace (lb DS/hr)	221.8	1,710.0
Stack flow rate (SCFM)	1,170	10,290

ELEMENTAL CONTAMINANTS

Wastewater sludges contain metals that could be hazardous if discharged to the atmosphere during sludge incineration. The effect of incineration on such metals will be influenced by the forms in which the metals are found in sludge. For example, if cadmium is present in the sludge in solution as cadmium chloride, it will volatilize during incineration. If present as a precipitated hydroxide, however, cadmium will probably decompose to the oxide, but will not volatilize at the temperatures of incineration. It is believed that most hazardous metals, with the exception of mercury, will not appear disproportionately in stack gases because of volatilization, but will instead be converted to oxides and appear as particulates either in the fly or bottom ash.

High temperatures during incineration decompose mercury compounds to volatile mercuric oxide or metallic mercury. Tests conducted on five incinerators equipped with scrubbers showed an average emission factor of 1.65 g of mercury emitted to the atmosphere per metric ton of dry sludge incinerated.

Table 61 (1017) presents concentrations of metals in the input sludge and the bottom ash for three plants. Any discrepancy in concentration would indicate volatilization of the metal in either the fly ash or vapor.

The chemical content of ash from multiple hearth incinerators reported by Olaxsey (1017) is shown in Table 62 .

TABLE 61. METAL TO FIXED SOLID RATIO FROM THREE INCINERATORS (1017) (mg/g)

Element	Plant A Sludge	Plant A Ash	Plant B Sludge	Plant B Ash	Plant C Sludge	Plant C Ash
Cadmium	0.37	0.20	n.d.	0.58	0.31	0.22
Chromium	2.0	0.3	2.9	0.5	0.7	0.6
Copper	2.6	1.3	2.5	1.7	1.6	1.6
Iron	18.0	8.9	12.0	11.0	50.0	43.0
Lead	5.8	0.7	7.0	0.85	2.0	1.0
Magnesium	9.0	n.d.	20.0	n.d.	6.0	n.d.

TABLE 62. CHEMICAL CONTENT OF SLUDGE INCINERATION ASH (1017)

Content	Percent of Total
Silica (SiO_2)	25-30
Alumina (Al_2O_3)	10-13
Iron Oxide (Fe_2O_3)	9-10
Magnesium Oxide (MgO)	2-2.8
Total Calcium (CaO)	30-37
Available Calcium	1-2
Phosphorus Pentoxide (P_2O_5)	7-10
Loss of Ignition	0.5-1.0

Chemical analysis of wastewater sludge ashes was also performed by Gray and Penessis (507). In general, the ashes were composed of predominantly silt-sized particles. All the ashes showed a pH in the range of 10.7 to 12.8 and were not appreciably soluble in water. The soluble salt content and conductivity of the ash leachates was almost exclusively composed of dissolved calcium and magnesium. Contrary to results on the two preceding tables, the amount of soluble iron, potassium, and phosphate was negligible, with the leachate containing between 0 and 0.5 mg/100 g of air-dried ash.

BIOLOGICAL PATHOGENS

Table 63 shows the effect of various pasteurization temperatures and times on the survival of typical pathogenic organisms found in sludge.

TABLE 63. EFFECT OF TIME AND TEMPERATURE ON THE SURVIVAL OF TYPICAL PATHOGENS FOUND IN SLUDGE* (1123)

Organism	Temperature °C				
	50	55	60	65	70
	-- minutes --				
Cysts of Entamoeba histolytica	5				
Eggs of ascaris lumbricoides	60	7			
Brucella abortus		60		3	
Corynebacterium diphtheria		45			4
Salmonella typhosa			30		4
Escherichia coli			60		5
Micrococcus pyrogene var. aursus					20
Mycobacterium tuberculosis var. promixis					20
Viruses					25

*Pathogens completely eliminated at indicated time and temperature.

ENVIRONMENTAL PATHWAYS

FRESH SURFACE WATER

INTRODUCTION

Direct discharge of treated wastewater to fresh surface water is the most popular method of wastewater disposal and potentially the most significant pathway of wastewater contaminants through the biosphere. In addition, relatively minor quantities of wastewater contaminants may indirectly reach fresh surface waters through runoff or percolation from land disposal of wastewater effluents and sludges. This section of the report discusses current knowledge about the fate of various contaminants in fresh-water systems.

Surface waters receive contaminants from treated municipal wastewaters (as previously stated) and from other significant sources such as agricultural runoff, storm-water discharges, mine drainage, and atmospheric fallout. Most of the literature pertinent to the behavior of various contaminants in fresh-water systems - listed in Table 64 - does not differentiate contaminants by source. This, however, does not affect the validity of the literature research.

Much of the material contained in this chapter was derived from the following references: 304, 403, 464, 636, 1126, 1266, and 1339.

WATER QUALITY PARAMETERS

General water quality parameters in surface waters are not of direct public health concern, although they often degrade the quality of the aquatic habitat. For example, phosphorus concentrations resulting from sewage disposal contributes to luxuriant growths of certain algae, such as Anabaena, Nodularia, or Nostoc-all of which produce toxins that can be harmful to humans (1331, 1472). However, the tastes and odor also associated with such water degradation would generally make the water unpotable long before the concentration of toxins reached levels harmful to public health.

Suspended solids in wastewater can carry adsorbed viral and other biological contaminants (1476). Trace metals occur in higher concentrations when associated with suspended matter than when they are in a dissolved state (227), a phenomenon that will be discussed more fully in the following section on elemental

TABLE 64. LITERATURE REVIEWED PERTAINING TO WASTEWATER TREATMENT PLANT EFFLUENT DISPOSAL TO FRESH-WATER SYSTEMS

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	46, 47, 54, 133, 138, 221, 364, 418, 454, 757, 829, 920, 1002, 1125, 1156, 1239, 1306, 1407, 1436, 1453
BOD	46, 133, 354, 364, 533, 757, 882, 1064, 1283, 1306, 1392, 1405, 1470
COD	46, 108, 133, 418, 1156, 1283,1306, 1470, 1475
Chlorides	46, 47, 54, 133, 500, 1018, 1156, 1177, 1306, 1405, 1470
Cyanides	1306, 1436
Fluorides	54, 133
Nitrates	9, 46, 47, 138, 151, 190, 221, 364, 411, 443, 454, 500, 533, 572, 600, 663, 757, 793, 829, 883, 1000, 1002, 1027, 1076, 1125, 1156, 1239, 1306, 1331, 1407, 1453
Nitrites	46, 133, 138, 221, 454, 663, 757, 793, 829, 920, 1125, 1156, 1239, 1407
Oil and grease	757, 1306, 1497
Phosphates	46, 47, 54, 133, 138, 190, 221, 364, 418, 454, 533, 555, 600, 663, 757, 784, 829, 882, 883, 1156, 1306, 1314, 1331, 1407, 1436, 1448, 1472
Suspended solids	47, 227, 354, 454, 757, 920, 1006, 1125, 1171, 1306, 1470, 1476
Total dissolved solids	364, 418, 454, 757, 1125, 1177, 1306
Total organic carbon	133, 1125, 1167, 1264, 1283, 1306
Other (general)	304, 336, 385, 403, 464, 757, 805, 1125, 1126, 1156, 1253, 1258, 1266, 1286, 1370

TABLE 64. (continued)

Contaminant	Reference Number
Elemental Contaminants	
Aluminum	28, 634, 636, 968, 1082, 1093, 1121, 1125, 1266, 1339, 1380, 1407, 1453
Antimony	303, 636, 777, 1093, 1121, 1266, 1306, 1339, 1380, 1453
Arsenic	303, 360, 409, 418, 520, 636, 1093, 1121, 1138, 1266, 1306, 1339, 1380, 1453
Barium	23, 418, 636, 968, 1093, 1121, 1266, 1339, 1380, 1453
Beryllium	418, 636, 1093, 1121, 1266, 1339, 1380, 1453
Boron	28, 133, 418, 636, 757, 1093, 1113, 1121, 1266, 1339, 1380, 1453
Cadmium	77, 117, 151, 175, 190, 263, 303, 360, 418, 433, 441, 462, 524, 629, 636, 724, 740, 757, 859, 1075, 1082, 1093, 1121, 1124, 1125, 1171, 1195, 1266, 1306, 1355, 1380, 1407, 1433, 1453, 1523, 1526, 1538
Chromium	151, 263, 365, 418, 473, 629, 777, 859, 1082, 1093, 1121, 1125, 1128, 1306, 1380, 1407, 1433, 1436, 1526, 1538
Cobalt	303, 445, 473, 600, 634, 636, 740, 782, 859, 968, 1056, 1074, 1075, 1082, 1093, 1121, 1125, 1266, 1339, 1380, 1456, 1538
Copper	28, 77, 151, 190, 206, 418, 433, 445, 451, 473, 504, 505, 629, 634, 740, 777, 782, 859, 968, 1044, 1074, 1075, 1082, 1093, 1121, 1125, 1171, 1266, 1339, 1355, 1407, 1433, 1453, 1454, 1543, 1562
Germanium	532, 636, 1093, 1121, 1266, 1339, 1453
Iron	28, 263, 306, 438, 473, 528, 634, 636, 782, 852, 854, 968, 1074, 1075, 1093, 1121, 1125, 1266, 1339, 1355, 1371, 1380, 1407, 1562

TABLE 64. (continued)

Contaminant	Reference Number
Lead	117, 151, 190, 204, 206, 360, 418, 441, 445, 462, 511, 524, 629, 634, 636, 724, 740, 757, 777, 782, 859, 912, 968, 1018, 1060, 1074, 1075, 1082, 1088, 1093, 1121, 1171, 1174, 1306, 1339, 1355, 1380, 1391, 1433, 1453, 1508, 1544, 1562
Manganese	28, 190, 206, 263, 306, 418, 445, 462, 473, 600, 636, 645, 844, 1074, 1075, 1121, 1171, 1266, 1306, 1339, 1355, 1380, 1433, 1453, 1523, 1538, 1544
Mercury	12, 37, 40, 78, 110, 117, 151, 269, 303, 418, 455, 462, 512, 524, 591, 607, 630, 634, 636, 726, 741, 752, 757, 760, 777, 809, 861, 968, 988, 1082, 1089, 1093, 1121, 1125, 1266, 1306, 1339, 1355, 1359, 1378, 1379, 1380, 1407, 1433, 1446, 1453, 1526, 1543, 1562
Molybdenum	303, 636, 1093, 1121, 1266, 1339, 1380, 1453
Nickel	28, 263, 418, 436, 445, 473, 629, 636, 782, 859, 968, 1074, 1093, 1121, 1125, 1171, 1266, 1306, 1355, 1380, 1407, 1453, 1544
Selenium	418, 636, 1093, 1114, 1121, 1266, 1306, 1339, 1380, 1453
Thorium	636, 1093, 1266, 1339, 1380, 1453
Tin	636, 1082, 1093, 1266, 1339, 1453, 1380
Uranium	636, 1093, 1121, 1266, 1339, 1380, 1453
Zinc	21, 77, 151, 190, 263, 303, 418, 441, 445, 504, 505, 524, 629, 636, 724, 740, 782, 844, 859, 968, 1074, 1075, 1082, 1093, 1121, 1125, 1171, 1266, 1306, 1319, 1339, 1355, 1380, 1407, 1433, 1453, 1523, 1538, 1544, 1562
Other (general)	133, 228, 636, 1211, 1339, 1380, 1454, 1508
Biocidal Contaminants	
Aldrin	139, 304, 373, 403, 783, 846, 1150, 1237, 1418

TABLE 64. (continued)

Contaminants	
Arsenated hydrocarbons	1237
Chlorinated hydrocarbons	139, 275, 304, 324, 373, 387, 403, 418, 464, 501, 512, 577, 757, 759, 772, 778, 827, 846, 998, 999, 1022, 1023, 1049, 1068, 1125, 1132, 1170, 1220, 1237, 1532, 1543
DDD	373, 403, 429, 577, 757, 759, 846, 1022, 1225
DDE	139, 373, 403, 429, 464, 529, 577, 759, 846, 972, 1125, 1150
DDT	14, 139, 304, 325, 373, 403, 420, 429, 453, 464, 515, 577, 641, 706, 757, 759, 846, 902, 931, 972, 1022, 1023, 1125, 1150, 1170, 1180, 1220, 1306, 1418, 1532
Dieldrin	373, 403, 429, 846, 972, 1125, 1550, 1237, 1306, 1418, 1470
Endrin	373, 403, 464, 846, 1150, 1418
Herbicides	597, 627, 1125, 1220, 1488, 1551
Organophosphorus pesticides	304, 306, 373, 418, 451, 501, 566, 846, 1196, 1245, 1550
Soil sterilants	73
Other (general)	291, 304, 418, 478, 600, 641, 827, 846, 846, 1049, 1090, 1225, 1550, 1454, 1512
Synthetic/Organic Contaminants	58, 149, 166, 275, 587, 770, 808, 856, 879, 998, 999, 1029, 1126, 1323, 1331, 1340, 1420, 1460, 1558
Biological Contaminants	
Adeno virus	757, 1509
Bacteria	74, 140, 197, 231, 243, 296, 304, 418, 434, 521, 588, 596, 656, 676, 678, 786, 815, 938, 1131, 1202, 1306, 1370, 1399
Clostridium botulinium	1131
Clostridium welchi	1131

TABLE 64. (continued)

Contaminants	Reference Number
Coliforms	46, 133, 140, 147, 187, 188, 197, 248, 299, 307, 354, 418, 457, 596, 612, 655, 656, 663, 757, 938, 1064, 1131, 1202, 1263, 1268, 1275, 1283, 1366, 1436, 1458, 1475, 1509, 1518
Coxsackie virus (A+B)	11, 588, 757, 1366, 1509
ECHO virus	11, 588, 757, 1366
Escherichia coli	147, 180, 588, 655, 1275
Fecal streptococci	140, 147, 180, 197, 307, 457, 458, 612, 656, 1131, 1202, 1263, 1283
Hepatitis virus	163, 257, 380, 492, 890, 1366, 1509
Leptospirosis	380
Polio virus	11, 85, 492, 750, 1209, 1366, 1509
Protozoa	655
Salmonella	30, 85, 231, 248, 307, 457, 588, 671, 1283
Shigella	30, 85, 588, 1449
Staphylococcus aureus	1131
Vibrio cholerae	74, 1331
Virus	46, 94, 99, 111, 133, 246, 251, 257, 380, 454, 468, 564, 588, 678, 750, 757, 815, 826, 877, 890, 938, 1182, 1207, 1237, 1240, 1283, 1331, 1366, 1434, 1443, 1478
Yeasts	596
Other (general)	180, 250, 433, 786, 1225, 1370

contaminants. Suspended matter and dissolved organics from wastewater can affect the natural biological community which, in turn, affects the nitrogen interconversions and nitrate concentrations.

The nitrogen-containing compounds (ammonia, nitrite, and nitrate) are theoretically hazardous because of nitrate's association with methemoglobinemia. Ammonia and nitrite nitrogen can be readily converted into nitrate by chemical or biological reactions. There are, however, no reported cases of detrimental public health effects resulting from nitrates in surface waters. This is because the dilution and natural processes occurring in surface waters prevent nitrate concentrations from reaching health impairing levels. There are, however, several reported cases of groundwater nitrate contamination rising to dangerous levels. These cases are discussed in the land/groundwater section of this report.

ELEMENTAL CONTAMINANTS

The behavior of elemental contaminants in fresh-water systems is very complex. Generally, elemental contaminant transport mechanisms can be divided into either (1) elements in solution, or (2) elements associated with inorganic or biological particulates. Each of these mechanisms can be broken down still farther. Dissolved elements may occur as unassociated ions or as inorganic or organic complexes. Elemental contaminant/inorganic particulate associations include coulombic attraction, as in conventional adsorption; ionic bonding, as in ion exchange; precipitated or coprecipitated metal coating; or incorporation into particulate crystalline lattices. Elemental contaminant/biological particulate associations include surface adsorption, ingested particulation, and biochemical incorporation into the organism. The particular transport mechanism that will predominate in a given water system depends, in part, on the geohydrologic environment, mineralogy/petrology of the river or lake bed, pH, temperature, dissolved organic or oxygen content, biological activity, elemental type and source, and nonelemental chemical composition of the water.

This variety of factors does much to explain the seeming discrepancies in the work of different researchers attempting to establish elemental distributions in fresh-water systems. For instance, Gibbs (473), in his examination of the Yukon and Amazon Rivers, concluded that precipitated metal coating and crystalline incorporation accounted for approximately 90 percent of the transported iron, nickel, copper, chromium, cobalt, and manganese (Table 65). Perhac (1074), on the other hand, in his analysis of two Tennessee streams, concluded that 95 percent of the total stream content of cadmium, cobalt, copper, nickel, lead, and zinc was in the dissolved state (Table 66). Assuming that there was no gross experimental error, widely differing environmental factors must have prevailed.

TABLE 65 . PERCENTAGES OF THE TOTAL AMOUNTS OF IRON, NICKEL, COBALT, CHROMIUM, COPPER, AND MANGANESE TRANSPORTED BY FIVE MECHANISMS IN THE YUKON AND AMAZON RIVERS (473)

Mechanism	Iron	Nickel	Cobalt	Chromium	Copper	Manganese
	Amazon River					
In solution and organic complexes	0.7	2.7	1.6	10.4	6.9	17.3
Adsorbed	0.02	2.7	8.0	3.5	4.9	0.7
Precipitated and coprecipitated	47.2	44.1	27.3	2.9	8.1	50
In organic solids	6.5	12.7	19.3	7.6	5.8	4.7
In crystalline sediments	45.5	37.7	43.9	75.6	74.3	27.2
	Yukon River					
In solution and organic complexes	0.05	2.2	1.7	12.6	3.3	10.1
Adsorbed	0.01	3.1	4.7	2.3	2.3	0.5
Precipitated and coprecipitated	40.6	47.8	29.2	7.2	3.8	45.7
In organic solids	11.0	16.0	12.9	13.2	3.3	6.6
In crystalline sediments	48.2	31.0	51.4	64.5	87.3	37.1

TABLE 66. HEAVY METAL DISTRIBUTION IN STREAMS (1074)

Average percentage of element occurring in specified form:

	Dissolved Solid	Coarse Particulate	Colloid
Cadmium	92.9	5.1	2.0
Cobalt	91.8	7.8	0.4
Copper	93.2	5.4	1.4
Iron	19.6	77.0	3.3
Lead	90.8	8.0	1.2
Manganese	23.4	74.6	2.0
Nickel	93.2	6.3	0.5
Zinc	77.8	21.8	0.4

The complexity of the chemistry, biology, and physics involved in water behavior of elements precludes a detailed discussion. Instead, a brief discussion is presented of the more important aspects of the behavior of elements in water, followed by a detailed examination of a few sample elements (mercury, arsenic, lead, cadmium, copper, iron) to demonstrate the principles involved.

Dissolved elements may occur as unassociated ions or as inorganic or organic complexes. Of the major elements under discussion in this report, only barium appears to any great extent as the unassociated cation; barium ions do not hydrolyze and form only weak complexes. However, several of the elements occur as unassociated anions: antimony, arsenic, boron, chromium, molybdenum, and selenium generally occur in fresh-water systems as the oxo anion. This is largely because the major sources of these elements, including wastewater, are rich in the anionic forms, and because the cationic forms are easily oxidized to the oxo anion in aquatic systems.

The majority of the dissolved elements normally exist as inorganic or organic complexes. Table 67 lists the more common ligands and the conditions and elements normally associated with them. In relatively pure water, aquo (H_2O) or hydroxo (OH^-) complexes are formed. At pH levels above neutral, many of the metal-hydroxo complexes are converted to metal hydroxides or oxides, which will precipitate out of solution or behave as colloids.

The other inorganic ligands responsible for keeping metals in solution in natural waters include carbonate, halides (notably chloride and fluoride), sulfur species (SH^-, sulfate, and sulfite), and nitrogen species (ammonia, nitrate, and nitrite). Most of the complexes formed from these ligands are thermodynamically unstable and appear as transition states between the free metal ion and a precipitate. The complexes, however, serve to keep the metals in solution for a time, and play a role in dissolving otherwise insoluble metals from precipitates or crystalline lattices.

Of somewhat more importance in terms of complex stability are the multidentate organic ligands. One of the reasons for the lack of inorganic complex stability is that most of the inorganic ligands are monodentate, i.e., there is one ligand for each metal ion coordination site. Organic ligands are multidentate, i.e., a given ligand can usually bond to two or more of a given ion's coordination sites. A multidentate complex (chelate) is more stable than a corresponding complex with monodentate ligands; thus, a chelate complex is apt to keep a metal ion in solution far longer than will an inorganic complex. Normally, this is not a problem; in relatively unpolluted fresh water the organic content is low, but in highly polluted water

TABLE 67. METALS COORDINATED BY LIGANDS NORMALLY FOUND IN NATURAL WATERS (1093)

Metal Carried	Ligands Responsible											
	H_2O	O^{-2} or OH^-	S^{-2} or SH^-	SO_4^{-2}	SO_3^{-2}	F^-	Cl^-	CO_3^{-2}	Organic	NH_3	NH_2OH	NO_2
Boron		1										
Aluminum	2	6				1	4	6	1			6
Barium	5			P				P(e)				
Chromium	2				6		4	6(e)	1	6		6
Copper	3		6				4(b)	6(e)	1	6	(d)	6
Cobalt	4		6				4	6(e)	1	6		6
Molybdenum		1	6					6				
Manganese	3						4	6(e)	1	6		
Iron	3		6		6	1	4	6(e)	1	6(c)		
Nickel	3		6				4	6(e)	1	6	(d)	
Zinc	3		6		6		4	6(e)	1	6	(d)	
Cadmium	4						4	6	1	6	1	6
Mercury	4(a)		6		6		1	6	1			
Lead	2			P			4	6(e)				6
Arsenic		1	6				4					
Antimony		1	6				4					

[1] Under all conditions found in normal natural waters (pH 0-11), this metal is coordinated by this ligand.

[2] Here, coordination occurs only at pH of less than about 4.

[3] Normally, coordination only occurs at a pH of less than 4; but, if the pCO_2 is low (high concentration), then bicarbonate which is water coordinated will be formed.

[4] Coordination occurs at a pH of 7 or less.

TABLE 67. (continued)

Footnotes

[5] The solubility falls markedly in the presence of this ligand at above pH due to precipitation of a carbonate or similar basic compound.

[6] Coordination occurs only at pH above 7 due to ligand instability, etc.

[7] Coordination occurs only at pH above 8-9.

[p] Precipitation almost always occurs. If nothing is marked, there is no coordination of this metal by this ligand in natural waters.

[a] Water will only coordinate if no other stronger ligand is present. In some cases, there is an equilibrium.

[b] Bromide and iodide resemble chloride except that they both precipitate silver, whereas silver chloride is fairly soluble due to $AgCl_2$-ions at high chloride concentrations. Iodide also precipitates copper and gold.

[c] Two valent iron in absence of air only.

[d] If ammonia is absent, a complex may be formed.

[e] Bicarbonate usually forms carbonate complexes, but metals so marked have a soluble bicarbonate which is water coordinated. Be and Tl have soluble water coordinated carbonates, and Ag has both sparingly soluble water coordinated carbonate and hydroxide.

unusually high soluble metal concentrations may result. Furthermore, there is evidence that some synthetic organic ligands, such as may be found in wastewater, form stronger complexes than do natural organic ligands (782). It has been demonstrated that one synthetic ligand, nitrilotriacetate (a proposed substitute for phosphates in detergents), is capable of dissolving significant quantities of precipitated lead out of bottom sediments (511, 1562).

The tendency for a given metal ion and ligand to form complexes depends on solution pH, concentration of the metal and ligand, concentrations of other metals and ligands in solution, equilibrium constants, redox conditions, and so forth. No two elemental contaminants behave exactly alike, and even different oxidation states of the same element may exhibit widely varying solution chemistry. Moreover, natural water systems are seldom in an equilibrium condition before, much less after, wastewater addition. This constantly changing system makes concise pathways almost impossible to construct. In general, low pH, an oxidizing environment, and the presence of a variety of ligands enhance solution tendencies. This is significant from a public health standpoint, because soluble element species are much more readily available for human contact than are precipitates or particulate elements.

Elemental contaminant/inorganic particulate interactions typically account for the bulk of the nondissolved element fraction. These interactions include coulombic or ionic attraction, precipitated or coprecipitated coating, or lattice incorporation. Coulombic attraction, or adsorption, is the least important transport mechanism except in the case of colloidal particulates, such as microparticulate iron or manganese hydroxides, which carry a weak negative charge and attract elemental cations. Larger particulates do not possess a strong enough charge to make coulombic attraction important.

Ionic attraction, or ion exchange, is somewhat more important. In this process, the heavy metals (Mn, Fe, Co, Ni, Cu, Zn, Mo, Cd, etc.) replace alkali and alkali earth cations (K, Na, Li, Mg, Ca) attached to crystalline lattices by ionic bonds. These ionic bonds will hold unless (1) the element is displaced by another element forming a stronger ionic bond, (2) a ligand forming a coordination bond stronger than the ionic bond breaks the ionic bond, or (3) an excess of alkali earth cations is available to force the equilibrium back to its original state.

According to Gibbs (473), lattice incorporation is generally insignificant as a transport or removal mechanism of dissolved wastewater elemental contaminants. It is a slow process that takes place in the sediments and has a more significant impact on the elemental composition of the sediments. The lattice-incorporated elemental burden of a water system is primarily from weathered rock, sand, and clay.

Precipitated and coprecipitated metal coatings account for most of the non-native particulate element content of a water system. Under favorable pH-EH conditions, metal precipitates will form. The initially small precipitate particles will tend to agglomerate or adhere to any available surface. In addition, cations held to particulates by coulombic or ionic bonds can form covalent bonds with anionic components of the particulate, if solution redox potentials change. In either case, the particulates are left with a coating of metal precipitates. This coating ultimately settles out of solution with the particulate, removing the elements from possible ready human contact unless they are redissolved by a change in redox conditions, a strong ligand, or some other agent strong enough to attach the precipitate. These coatings may account for the bulk of the inorganic particulate element burden of wastewaters.

Elemental contaminants incorporated with biological particulates constitute the remainder of the particulate elemental burden of a water body or wastewater. This incorporation may take the form of surface adsorption, inorganic particulate ingestion, or biochemical incorporation into an organism's tissues. Obviously, the role played by biological transport is highly dependent on the type and quantity of organisms present. Surface adsorption is usually associated with microorganisms. Particulate ingestion is associated with those organisms that have internal digestive organs (e.g., fish, crustaceans, worms, etc.); ingested particulates are usually eliminated within a short time, but the elements associated with them may be biochemically incorporated into the organism's tissues. From a public health standpoint, soluble and biochemically incorporated elemental contaminants are the most important, for it is by these routes that potentially hazardous elements reach man. Biochemical incorporation involves both essential trace element concentration (e.g., cobalt in vitamin B-12) and reaction of an element with cellular chemicals (e.g., the reaction of mercury with sulfur-containing amino acids in proteins). Both plants and animals are involved and, thus, the concentration of a given element may move up the food chain. The incorporation is reversible; once the organism in its environment is removed from contact with the element, the latter can gradually be excreted.

No two elements behave exactly alike; furthermore, the number of factors available that can affect transport mechanisms makes the possibilities nearly endless. There are similarities, however, that make the use of examples illustrative and useful; mercury, arsenic, iron, cadmium, copper, and lead will be used to provide detailed descriptions of the general pathways disucssed above.

Mercury from wastewater enters a water system primarily as the metal or divalent cation. Although of limited solubility, it can reach concentrations of 100 ppb in aerated water (455). Metallic mercury alone is soluble up to 25 ppb and will hydrolize to soluble Hg $(OH)_2$ in oxygenated systems, increasing the overall solubility and water content. Despite these solubility figures, mercury concentrations seldom exceed 5 ppb (Table 68) except in polluted water.

Inorganic and biological adsorption, absorption, and precipitation serve to keep the concentrations of dissolved mercury much lower than the theoretical maximum. In general, the bulk of the mercury in a given water system is in the sediments; Table 69 gives a summary of some of the factors affecting mercury incorporation in the sediments. In reducing sediments, mercury is tied up as the sulfide, although if the system becomes sufficiently alkaline, HgS_2 may be released into solution. Should the sediments become aerobic, the sulfide will be oxidized to sulfate, and the mercury will be released.

All soluble mercury species except mercuric sulfide can be absorbed by bacteria. Once the mercury is in the bacteria, a series of transformations - possibly via a detoxification mechanism - convert the incorporated mercury into mono- and dimethyl mercury, both soluble at low concentrations and readily released into solution. The methyl mercury compounds are much more lipid-preferring than the inorganic forms and are quickly absorbed by living tissues. As a rule, mercury concentrations tend to increase in organisms up the food chain, so that the highest concentrations are found in fish. This is partly due to absorption of methyl mercury from the water and partly from ingestion of plants or smaller organisms containing methyl mercury. When the organisms die, the mercury returns to the sediments, where most of the bacterial methylation occurs. Table 70 lists some sample sediment and plankton/algae mercury concentrations in Lake Erie.

Arsenic, selenium, and antimony are chemically similar and exhibit analogous environmental behavior. Arsenic has been studied far more than either selenium or antimony. The following discussion of arsenic is largely applicable to selenium and antimony as well.

Arsenic has an unusually complex chemistry in aquatic systems: oxidation-reduction, ligand exchange, precipitation, adsorption, and biomethylation reactions all take place. Arsenic species can be removed from water via surface adsorption and coprecipitation with metal ions; both arsenate (AsO_4^{-3}) and arsenite (AsO_3^{-3}) have a high affinity for hydrous iron oxides and readily coprecipitate with or adsorb onto them. Significantly, iron ores are always enriched with arsenic (409). Aluminum hydroxide and clays are adsorb arsenate species, although to a lesser degree.

TABLE 68 . SELECTED CONCENTRATIONS OF MERCURY IN NATURAL WATERS (455)

Source and Location	Mercury (ppb)
River water, European USSR	0.4-2.8
River water, Armenia	1-3
Saale River, Germany	0.035-0.145
River water, Italy	0.01-0.05
River water, near mercury deposits, Italy	up to 136
Colorado River, Arizona	<0.1
Ohio River, Illinois	0.1
Mississippi River, Kentucky	<0.1
Missouri River, Montana	<0.1
Missouri River, St. Louis, Missouri	2.8
Kansas River, Topeka, Kansas	3.5
Hudson River, New York	0.1
Lake Champlain, New York	<0.1
Maumee River, Antwerp, Ohio	6.0
Delaware River, New York	<0.1

TABLE 69. FACTORS INFLUENCING THE DEGREES OF ABSORPTION OF Hg^{++} TO THE RIVER SEDIMENTS* (861)

Sediments	Conditions	Final pH	% Remained in water	% Absorbed to sediment	% Organic ** mercury
Natural samples					
Chippewa Falls	control +	6.5	6.0	89.2	4.8
Wisconsin River	low Hg	7.0	8.9	83.7	7.4
	medium Hg	7.25	44.3	49.9	5.8
	high Hg	7.10	40.4	47.6	12.0
Chippewa Lake	low Hg	6.0	19.5	68.1	12.4
Anaerobic condition					
Chippewa Falls	N_2	7.7	13.6	74.4	12.0
pH artificially adjusted					
Chippewa Falls	HCl	3.95	3.0	92.3	4.7
	HCl	4.95	3.0	92.4	4.6
	HCl	6.10	6.6	88.6	4.8
	Na_2CO_3	7.15	2.5	93.1	4.4
	Na_2CO_3	8.05	19.9	76.0	4.1
	Na_2CO_3	8.95	60.0	35.0	4.1
Chemicals added					
Chippewa Falls	EDTA (10^{-3}M)	6.4	44.6	49.2	6.2
	FAD (10^{-3}M)	6.5	22.4	74.9	2.7
	GSH (2 x 10^{-5}M)	6.1	17.4	78.5	4.1
	Glucose (10^{-2}M)	5.8	13.6	82.7	3.7
	H_2S (gas)	6.3	6.0	79.9	14.1
	$SnCl_2$ (10^{-4}M)	5.8	1.9	92.5	5.6

* Results shown by percentages of originally added $203_{Hg}{}^{++}$ (10^{-8} mole) recovered in each fraction.

** Toluene extractable mercury compounds from both water and sediments.

+ The sample served as the control for all subsequent tests.

TABLE 70. MERCURY CONTENT OF SEDIMENTS AND PLANKTON/ALGAE SAMPLES COLLECTED FROM LAKE ERIE (1089)

Station No.	Approximate Location	Mercury content in µg/g*	
		Sediments**	Plankton/Algae
01	Buffalo River	2.0	31.2
02	Cattaraugus Creek	1.2	25.1
03	Barcelona	0.6	2.8
04	Ashtabula	4.6	7.4
05	Fairport	1.5	12.8
06	Cleveland	12.0	33.5
07	Toledo	10.4	20.5
08	Detroit River	4.5	26.1
09	Mid. Bass Island	1.5	20.1
10	Port Crewe	0.5	12.4
11	Port Stanley	1.5	12.0
12	Long Point	7.0	14.7
13	Long Point Bay	1.0	23.7
14	Port Maitland	1.8	15.4
15	Mid-Lake	1.5	0.6
16	Black Rock Channel	12.4	27.8

* In terms of the equivalent dry wt of the sample.

** Sediment samples from 3 to 30 cm below the water-sediment interface.

Microbial transformations of arsenic, while demonstrable in the laboratory, have not been positively identified in natural water systems. The two most commonly postulated transformations are oxidation of arsenite and methylation. Methylation is important because it could be a means by which sediment arsenic is recycled back into the water system; natural aquatic methylation has not been demonstrated.

Soluble iron entering a water system in wastewater will usually be either ferrous (Fe II) or complexed ferric (Fe III) iron. The former is much more soluble than the latter (which has a stronger tendency to form complexes), although neither tends to remain in solution long. In the surface layers of most natural waters, pH levels and oxygen conditions are such that Fe (II) is readily oxidized to Fe (III), which just as readily hydrolyzes to insoluble hydrous ferric oxide (FeOOH). Hydroxide has a much stronger affinity for ferric iron than do basic organic or inorganic ligands.

Hydrous ferric oxide tends to form microcrystalline precipitates of a colloidal nature, so that it is almost impossible to analytically distinguish between soluble and colloidal iron. Consequently, the two forms of iron are usually reported together as soluble iron. Although hydroxide supersedes other anionic ligands, frequent incorporation of coordinating anions into ferric oxide precipitates enhances colloidal stability and further blurs the distinction between the colloid and soluble ferric complexes.

Cadmium readily precipitates as the hydroxide or carbonate and consequently is not normally found in high concentrations in surface waters. In fact, several researchers (360, 968) have noted that high soluble cadmium concentrations are invariably associated with polluted water that receives a steady cadmium source, such as industrial wastewater.

Cadmium (II) readily hydrolyzes and forms transitory inorganic complexes, such as chloride complexes that have a limited affinity for hydrous iron and manganese oxides, and organic particulates. The organic affinity probably indicates a reaction between the cadmium and sulfur-containing compounds.

Cadmium forms the insoluble hydroxide at pH levels of 7 and above; it forms the insoluble carbonate under oxidizing conditions, particularly in soft waters where cadmium does not have to compete with calcium and magnesium for the carbonate anion. Once cadmium has precipitated and settled into the sediments, it is not readily removed. Consequently, if cadmium additions are reduced, a water body will tend to purify itself of soluble cadmium.

Copper, and to a lesser extent nickel, occupy an unusual position in water chemistry and biology because they are both nutrients and toxins. This has a pronounced effect on their water chemistry. Copper contained in wastewater may be either soluble or particulate; neither form predominates as a rule. Copper adsorbs readily onto clay and organic particulates. Copper also forms several very stable complexes. In pure water, while the aquo complex may predominate, the carbonate, chloride, and amine inorganic complexes are much more stable.

Ultimately, the stability soluble of copper can be attributed to organic complexes, since copper forms coordination complexes with virtually every conceivable organic ligand. These complexes are very stable thermodynamically and are also resistant to microbial attack, a mechanism responsible for the destruction of most organic complexes. Copper is a bacterial toxin and, if released from its complex by microbial attack, simply kills the offending bacteria and forms a new complex (968).

Copper is removed from solution via precipitation or biological incorporation. Since the most common precipitate is the carbonate, most sediment copper is in the carbonate form (263, 968). An essential trace nutrient, copper is readily incorporated into aquatic plants and animals.

The main soluble species of lead in wastewater are the lead (II) cation and the hydrolyzed complex $Pb\ (OH)_3^-$. Lead forms a variety of stable complexes as well; researchers (777) have identified both $PbOH^+$ and $Pb(CO_3)_2^{-2}$ in natural water systems. Lead complexes easily with a variety of organic chelates, forming very stable complexes. Some of these complexes are more stable than the precipitated lead in the sediments; therefore, they will actually dissolve otherwise insoluble lead. A case in point is nitrilotriacetate, which can solubilize lead from lead carbonate precipitates (511).

Low water organic content generally prevents solution lead concentrations from exceeding a few parts per billion. In most water systems, lead introduced with wastewater readily forms insoluble $Pb(OH)_2$ and $PbCO_3$, which will precipitate and adsorb onto suspended particulates. Ionic lead is not so strongly adsorbed, although it does have some affinity for clays.

Hydrous iron oxides strongly sorb ionic lead at neutral to slightly acidic pH levels. Some ionic or complexed lead adsorbs onto or is chelated with the surface mucilage of algae, and microorganisms immobilize substantial quantities of inorganic lead, presumably on or in all membranes (1391). As a result of all of these adsorption/precipitation mechanisms, most of the water lead burden is associated with particulate matter, and most of the lead entering a normal water system ultimately finds its way into the sediment.

Natural water bodies normally contain very low dissolved concentrations of the more harmful elemental contaminants. Unless wastewater additions are voluminous and repeated, natural water chemistry can purify the water of soluble species fairly well. However, there is a buildup of these elementals in the sediments. This means that if the local water chemistry should change significantly, the elements can still be released to solution. Natural water purification mechanisms can only change the wastewater elemental problem from a real to a potential hazard; they cannot solve the problem of elemental contamination.

BIOCIDAL CONTAMINANTS

The single most important source of biocidal contaminants in fresh-water bodies is surface runoff, followed by aerial fallout and industrial waste discharge from plants manufacturing biocides. In general, municipal wastewater will have detectable quantities of biocides only if it contains biocide manufacturing wastes. Cleanup and disposal by households, farmers, gardeners, etc., contribute minimally to the overall wastewater burden.

In this discussion, biocides will include chlorinates, hydrocarbons, organophosphates, carbamates, and ionic biocides (Table 71).

Biocides can be transported or removed from a system by microbial or chemical degradation, photodegradation, sediment or humic matter, adsorption, volatilization, and biological uptake. All of these mechanisms are in turn affected by pH, temperature, salt or organic content, and bioproductivity. One mechanism that has not been studied to any great degree - especially in fresh water - is aerosolization. This mechanism will be discussed in greater detail in the marine water section of this chapter, as its importance in fresh-water systems is limited mainly to the larger lakes. Briefly, however, aerosolization occurs through the action of wind and waves on floatables. Aerosols and particulates can be released to the air and transported great distances by the wind; biocides can concentrate in floatables via dissolution in surface oil, adsorption on floating matter, or flotation (caused by low specific gravity and insolubility), thus becoming amenable to the aerosolization process. As mentioned above, this is not an important transport mechanism in most fresh-water systems.

Various transport mechanisms affect the biocide classes differently. This is demonstrated in Table 72 , which compares the persistence of selected chlorinated hydrocarbons. organophosphates, and carbamates in river water Because of these differences, the biocide classes will be discussed separately.

TABLE 71 . BIOCIDE TYPES AND EXAMPLES

Chlorinated Hydrocarbons

- DDT (DDD, DDE)
- Methoxychlor
- Endrin
- Dieldrin
- Aldrin
- Toxaphene
- Lindane
- Chlordane
- Heptachlor

Organophosphates

- Parathion
- Malathion
- Dimethoate
- Methyl parathion
- Phorate
- Demeton
- Ethion
- Disulfaton

Carbamates

- Carbaryl
- Sevin
- Baygon
- Pyrolan
- Dimetilan

Ionic Biocides

- Diquat
- Paraquat
- Chlormequat
- Morfaunquat
- Phosphon
- Hyamine
- 2,4-D
- 2,4,5-T
- Dalapon
- Silvex
- Dichlobenil

TABLE 72. PERSISTENCE OF COMPOUNDS IN RIVER WATER (846)

Compound	Original compound found, percent				
	0-time	1 wk	2 wk	4 wk	8 wk
Organochlorine compounds					
BHC	100	100	100	100	100
Heptachlor	100	25	0	0	0
Adlrin	100	100	80	40	20
Heptachlor epoxide	100	100	100	100	100
Telodrin	100	25	10	0	0
Endosulfan	100	30	5	0	0
Dieldrin	100	100	100	100	100
DDE	100	100	100	100	100
DDT	100	100	100	100	100
DDD	100	100	100	100	100
Chlordane (tech.)	100	90	85	85	85
Endrin	100	100	100	100	100
Organophosphorus compounds					
Parathion	100	50	30	<5	0
Methyl parathion	80	25	10	0	0
Malathion	100	25	10	0	0
Ethion	100	90	75	50	50
Trithion	90	25	10	0	0
Fenthion	100	50	10	0	0
Dimethoate	100	100	85	75	50
Merphos	0	0	0	0	0
Merphos recov. as Def	100	50	30	10	<5
Azodrin	100	100	100	100	100
Carbamate compounds					
Sevin	90	5	0	0	0
Zectran	100	15	0	0	0
Matacil	100	60	10	0	0
Mesurol	90	0	0	0	0
Baygon	100	50	30	10	5
Monuron	80	40	30	20	0
Fenuron	80	60	20	0	0

The chlorinated hydrocarbon pesticides are all insoluble in water, with the exception of lindane, which is sparingly soluble to 10 ppm (403). They are generally resistant to microbial and chemical degradation, as evidenced by their estimated environmental half-lives, shown in Table 73 .

TABLE 73 . ESTIMATED PESTICIDE HALF-LIVES (403)

Pesticide	Half-Life, yrs
Lindane	2
Chlordane	8
Toxaphene	11
Heptachlor	2 to 4
DDT	10 to 20
Endrin (Dieldrin)	8 to 10

These pesticides are somewhat more susceptible to photodegradation, although the degradation products are often as toxic as the parent compound, regardless of the type of degradation. DDT is decomposed chemically to DDD and DDE and photochemically to PCB's (304, 1170); aldrin is photooxidized to the more toxic dieldrin (304); and methoxychlor is degraded to methoxychlor DDE (1049). Surface oil slicks tend to concentrate chlorinated hydrocarbons and thus make them more available for photochemical degradation (304).

Chlorinated hydrocarbons in general readily adsorb onto fungi, algae, and floc-forming bacteria (783, 1049), and thus tend to concentrate in biological communities. When ingested by higher organisms, they accumulate in lipid tissues; consequently, there is a tendency for chlorinated hydrocarbons to concentrate up the food chain.

Chlorinated hydrocarbon insecticides differ in chemical structure, but they all exhibit affinity for organic sediments and resistance to microbial attack. As a result, there is accumulation in bottom sediment. Research on Lake Michigan demonstrates this, as shown in Table 74 . Routh (1180) showed that DDT, with its affinity for fine particulate clay sediments, concentrated up to 20 times normal background levels, from 10 to 200 ppb. This affinity for organic matter and particulates leads to high sediment DDT concentrations. Table 75 shows DDT concentrations in stream sediments over a period of time. Adsorption of DDT on algae can be 10 to 100 times greater than adsorption on clay (706). Moreover, DDT seems to have an inhibitory effect on sediment bacteria (14).

There has been little research on other chlorinated hydrocarbons, much of it limited to an evaluation of environmental

TABLE 74. CHLORINATED HYDROCARBON INSECTICIDES IN SOUTHERN LAKE MICHIGAN SEDIMENTS (μg/ℓ) (778)

Insecticide	Top interval 0-2 cm			2-6 cm			6-12 cm		
	Median	50% Mid-range	No. of Samples	Median	50% Mid-range	No. of Samples	Median	50% Mid-range	No. of Samples
p,p'-DDT	9.3	5.5-17	59	3.8	2.6-5.2	40	3.0	1.5-6.0	20
o,p-DDT	1.2	Tr-2.0	54	0.7	Tr-1.0	37	Tr	ND-2.3	14
p,p'-DDE	2.2	0.6-3.5	59	0.8	Tr-1.5	40	0.6	Tr-2.1	19
o,p-DDE	Tr	Tr	49	ND	ND	32	ND	ND-Tr	12
p,p'-DDD	3.0	1.4-10	54	0.5	Tr-2.0	37	ND	ND	12
o,p-DDD	Tr	Tr-1.8	49	ND	ND-0.5	32	ND	ND	12
Total DDT complex	18.5	10-32	59	6.3	3.9-13	40	3.4	2.2-8.1	20
Dieldrin	2.0	1.3-4.1	54	Tr	Tr-1.1	37	Tr	Tr-0.9	14
Heptachlor epoxide	Tr	Tr-0.7	54	Tr	ND-0.5	32	ND	ND	14
Lindane	Tr	Tr	45	ND	ND-Tr	30	ND	ND	14

TABLE 75. DDT CONCENTRATIONS IN STREAM SEDIMENTS (515)

Years after one application	DDT (ppm) muds
0	.83 (.04-1.6)
1	1.08 (.25-1.9)
2	--
3	.21 (.12-.30)
4	.21 (.16-.25)
5	.59 (.07-1.9)
6	.06 --
7	.21 (.11-.31)
8	.13 --
9	.03 (.02-.04)
10	.07 (0-.16)
Never sprayed	.006 (0-.02)
Two sprays	.43 (.17-.77)
Three sprays	.35 (.17-.6)

levels. Table 76 gives the results of one such survey for dieldrin.

TABLE 76 . DIELDRIN IN RIVER BOTTOM SILTS (972)

Source	Dieldrin (ppb)
Iowa River	8.8
Des Moines River	35
East Nishnabotna River	21
West Nishnabotna River	16
Upper Iowa River	<1
Johnson County Creek	170

Other researchers have found that the highest reported concentrations of several pesticides in major U.S. river basins from 1958 to 1965 were as follows: dieldrin, 0.100 μg/ℓ; endrin, 0.116 μg/ℓ; and DDT, 0.148 μg/ℓ. Dieldrin was the most widely found pesticide (1470).

The organophosphorus biocides are more soluble than the chlorinated hydrocarbons. These solubilities range from 1 ppm for ethion to 20,000 ppm for dimethoate; most fall in the 25 to 150 ppm range (403). The organophosphorus biocides are also more amenable to both microbial and chemical degradation. Even parathion, the most chemically resistant of the organophosphates, will degrade via ester linkage hydrolysis in a few months under normal conditions. The degradation takes place in just a few weeks in polluted water with a high bacteria count (501). Yu and Sanborn's (1550) experimental evaluation of parathion in a model ecosystem yielded a calculated half-life of 15 to 16 days. In a similar study, guthion yielded a half-life of one month at pH levels less than 9 and a half-life of less than one week at more alkaline pH's.

Interestingly, the degradation of organophosphates can be inhibited by the presence of other synthetic organic chemicals. Experiments were conducted with two detergent surfactants - alkyl benzene sulfonate (ABS) and linear alkyl benzene sulfonate (LAS). These experiments demonstrated increased persistence for several organophosphate insecticides, especially parathion and diazinon (304). As a result, highly polluted water may exhibit accumulations or half-lives far beyond the normal for organophosphates, which, as a rule, neither persist nor accumulate in the environment, but are removed entirely within a few months.

Carbamate biocides are moderately soluble, ranging from 7 ppm for terbutol to 250 ppm for propham and averaging around 100 ppm (403). In general, they decompose easily and show little tendency toward adsorption on suspended material, but hydrolyze readily. The hydrolysis is particularly pH dependent, virtually ceasing below pH 5 (403) and increasing as the pH and temperature rise. High salt content affects the hydrolysis rate inversely, slowing the rate as the salt concentration increases (403). Carbamates photodecompose readily - increasingly so, as the pH rises - and can be rapidly biodegraded under normal circumstances (1170). Carbamates are not, then, persistent in normal water systems, lasting only a few days to a few weeks, but remain as a stable compound in acidic waters (403).

Ionic biocides are a broad class embracing a variety of chemical types and uses. They are all considered soluble in water, with solubilities ranging from 100 to more than 1,000,000 ppm. Ionic biocides that are marginally soluble in pure water have increased solubilities in natural waters high in humic acid salts (403). With few exceptions, these biocides do not accumulate or persist and, consequently, are seldom found in high concentrations.

Ionic biocides are, however, strongly adsorbed onto soil particles, all types of clay, humic matter, and organisms - in short, onto anything with a partial charge or an ion exchange capability (403). They are generally resistant to chemical attack but photodegrade readily, except when adsorbed onto particulate matter (403). Ionic biocides respond differently to microbial attack, but are absorbed by many organisms. As a result, they tend to concentrate in organisms and up the food chain. Research on TCDD, an ionic herbicide residue, demonstrated that accumulation was directly related to water concentrations (0.05 to 1,330 ppb) and averaged between 400,000 and 2,000,000 times the water concentration (627).

SYNTHETIC/ORGANIC CONTAMINANTS

Recently, there has been a great interest in identifying synthetic/organic trace compounds in water supplies drawn from rivers and in other water bodies receiving treated wastewater. Although studies have been made of the concentrations found in various water systems, neither the environmental pathways nor the potential health effects to man of these substances have been studied to any great extent.

Over 100 synthetic/organic compounds have been identified in various drinking water sources. Thirty-six compounds were found in the lower Tennessee River (Table 77), while 66 were identified and quantified in Mississippi River water at New Orleans (Table 78). Table 79 lists the results of organic analyses of several other domestic water supply sources.

The many different types of compounds under discussion here make generalizations difficult regarding their environmental

TABLE 77. ORGANIC COMPOUNDS IDENTIFIED UP TO 1975 IN LOWER TENNESSEE RIVER (1323)

COMPOUNDS

- Acenaphthene
- Allylbenzoate
- Anthracene
- Benzene
- Biphenyl
- Butylbenzene
- 5-Chloro-2-Methylbenzofuran
- p-Cresol
- Diallyl Adipate
- Dibutyl Phthalate
- Diphenylacetylene
- 1,1-Diphenylethene
- 2,6-Di-Tert-Butyl-4-Methylphenol
- Ethylbenzene
- Ethyl o-Phthalate
- Ethylstyrene
- Ethylene Dimethylacrylate
- Fluoranthene
- Fluorene
- Hexachlorobenzene
- Indene
- o-Methoxybenzoic Acid
- 2-Methylanthracene
- 2-Methylbiphenyl
- 4-Methyldiphenylacetylene
- Methyl Indene (2 isomers)
- 1-Methylnaphthalene
- Naphthalene
- p-Nonylphenol
- n-Octyl-o-Phthalate
- Pyrene
- Styrene
- 1,2-Tetradecanediol
- Toluene
- 3,4,4-Trimethyl-2-Hexene
- Xylene

TABLE 78.

ORGANIC COMPOUND IDENTIFICATIONS

NEW ORLEANS AREA WATER SUPPLY STUDY
(856)

	Compound	Highest Measured Concentration µg/1		
		Carrollton Water Plant	Jefferson # 1 Water Plant	Jefferson # 2 Water Plant
1	Acetaldehyde	D-VOA	NE	NE
2	Acetone	D-VOA	NE	NE
3	Alkylbenzene-C_2 isomer	0.05	ND	ND
4	Alkylbenzene-C_2 isomer	0.33	ND	ND
5	Alkylbenzene-C_2 isomer	0.11	0.03	ND
6	Alkylbenzene-C_3 isomer	0.01	ND	ND
7	Alkylbenzene-C_3 isomer	0.04	0.05	0.02
8	Alkylbenzene-C_3 isomer	0.02	ND	ND
9	Atrazine * (2-chloro-4-ethylamino-6-isopropylamino-*s*-triazine)	5.0	4.7	5.1
10	Deethylatrazine (2-chloro-4-amino-6-isopropylamino-*s*-triazine)	0.51	0.27	0.27

TABLE 78. (continued)

	Compound	Highest Measured Concentration µg/l Carrollton Water Plant	Jefferson # 1 Water Plant	Jefferson # 2 Water Plant
11	Benzyl butyl phthalate*	0.64	0.81	0.73
12	Bromodichloroethane	D-VOA	NE	NE
13	Bromoform *	0.57	ND	ND
14	Butanone	D-VOA	NE	NE
15	Carbon disulfide	D-VOA	NE	NE
16	Carbon tetrachloride	D-VOA	NE	NE
17	bis-2-Chloroethyl ether*	0.07	0.16	0.12
18	Chloroform *,a	113	NE	NE
19	bis-2-Chloroisopropyl ether *	0.18	0.05	0.03
20	n-Decane *	0.04	ND	ND
21	Decane-branched isomer	0.03	ND	ND

TABLE 78. (continued)

	Compound	Highest Measured Concentration µg/l Carrollton Water Plant	Jefferson # 1 Water Plant	Jefferson # 2 Water Plant
22	Dibromodichloroethane isomer	0.33	ND	0.63
23	Dibromochloromethane *	1.1	0.30	0.60
24	Dibutyl phthalate *	0.10	0.16	0.19
25	2,6-Di-t-butyl-_p_-benzoquinone *	0.22	0.19	0.23
26	Dichlorobenzene isomer	0.01	D-RE	ND
27	1,2-Dichloroethane [a]	8	NE	NE
28	Dichloromethane	D-VOA	NE	NE
29	Dieldrin **	0.05	0.07	0.05
30	Diethyl phthalate *	0.03	0.03	0.01
31	Di(2-ethylhexyl) phthalate *	0.10	0.31	0.06
32	Dihexyl phthalate	0.03	ND	ND

TABLE 78. (continued)

		Highest Measured Concentration µg/l		
	Compound	Carrollton Water Plant	Jefferson # 1 Water Plant	Jefferson # 2 Water Plant
33	Dihydrocarvone	0.14	0.06	0.07
34	Diisobutyl phthalate *	0.59	ND	ND
35	Dimethyl phthalate	0.27	0.13	0.18
36	Dioctyl adipate	0.10	ND	ND
37	Dipropyl phthalate *	0.07	0.13	0.14
38	n-Dodecane *	0.01	ND	ND
39	Endrin **	0.004	NYE	NYE
40	Ethanol	D-VOA	NE	NE
41	_o_-Ethyltoluene *	ND	0.04	0.02
42	_p_-Ethyltoluene *	0.02	0.03	0.03
43	1, 2, 3, 4, 5, 7, 7-Heptachloronorbornene *	0.06	0.05	0.05

TABLE 78. (continued)

	Compound	Highest Measured Concentration µg/l Carrollton Water Plant	Jefferson # 1 Water Plant	Jefferson # 2 Water Plant
44	Heptachloronorbornene isomer	0.06	0.04	0.04
45	Hexachloro-1,3-butadiene *	0.16	0.27	0.21
46	Hexachloroethane *	4.4	0.19	0.16
47	Isophorone *	1.5	2.2	2.9
48	Limonene *	0.03	ND	ND
49	Methanol	D-VOA	NE	NE
50	Methylbenzoate	ND	D-RE	ND
51	3-Methylbutanal	D-VOA	NE	NE
52	2-Methylpropanal	D-VOA	NE	NE
53	n-Nonane *	0.03	ND	ND
54	n-Pentadecane *	0.02	ND	ND

TABLE 78. (continued)

	Compound	Highest Measured Concentration µg/l		
		Carrollton Water Plant	Jefferson # 1 Water Plant	Jefferson # 2 Water Plant
55	Tetrachloroethane isomer	0.11	ND	ND
56	Tetrachloroethylene	D	0.5	0.41
57	n-Tetradecane *	0.02	ND	ND
58	Toluene *	0.08	0.10	ND
59	1,1,2-Trichloroethane *	0.35	0.45	0.41
60	1,1,2-Trichloroethylene	D-VOA	NE	NE
61	n-Tridecane *	0.01	ND	ND
62	Trimethyl-trioxo-hexahydrotriazine isomer	0.07	ND	ND
63	Triphenyl phosphate *	0.12	ND	ND
64	n-Undecane *	0.02	ND	ND
65	Undecane-branched isomer	0.04	ND	ND
66	Undecane-branched isomer	0.06	ND	ND

TABLE 78. KEY TO SYMBOLS USED IN TABLE
KEY TO SYMBOLS USED IN TABLE

Symbols used in column headed Compound

*	While all compounds listed in the table were identified by one or more methods, those marked with this symbol gained added confirmation by gas chromatography retention time match with an available standard of the compound.
**	Compounds marked with this symbol gained further confirmation by gas chromatography retention time match with available standards on each of three different columns, polar and non-polar.
a	The quantitative values for these compounds were obtained on Volatile Organics Analysis by comparison with standards of known concentration at the Water Supply Research Laboratory. Compound 18 was detected but not quantified in Tetralin extracts of Carrollton water at Southeast Environmental Laboratory, but not in Tetralin extracts of Jefferson No. 1 or Jefferson No. 2. The latter laboratory did not detect compound 27.

Symbols used in columns headed Highest Concentration Measured.

D-VOA	These compounds were detected by Volatile Organics Analysis - Bellar Technique only. Quantitative values have not yet been obtained. This method was performed only on the Carrollton water at the Water Supply Research Laboratory.
D-RE	These compounds were detected only on XAD resin extracts in the specific water for which this symbol is used. Quantitative values were not obtained from the resin extracts. The compound may have been detected and quantified by another method in one or both of the other waters examined.
D	In the one instance where this symbol was used the compound was detected by both the Water Supply Research Laboratory and Southeast Environmental Research Laboratory but not quantified by either laboratory.
NE	This symbol means not examined. It is used exclusively for some compounds reported by the Water Supply Research Laboratory. This laboratory did not obtain samples of water from Jefferson No. 1 or Jefferson No. 2.

TABLE 78. (continued)

ND	This symbol means the compound was not detected in that specific water by any of the methods employed.
NYE	Compound 39 was confirmed in Carrollton water carbon chloroform extracts shortly before preparation of this report. Jefferson No. 1 and Jefferson No. 2 extracts have not yet been re-examined specifically for compound 39.

TABLE 79. MOLECULAR CONSTITUENTS IDENTIFIED IN NATURAL WATER SAMPLES (1323)

Constituent	Sample*	Concentration mg/ℓ
p-Cresol	3	7
Diethylene glycol	5	1
Ethylene glycol	5	20
Glycerine	1,2,3,4,5	1-20
Glycine	1	2
Mannitol	5	2
Methyl-a-D-glucopyranoside	4	30
Methyl-B-D-glucophranoside	4	3
Sucrose	1,5	2
Xylitol	5	1
Urea	1,2	4
Inositol	1,2,3,4,5	0.5-1
O-Methylinositol	1,2,3,4,5	0.3-10
Linoleic Acid	1,5	1
Oleic Acid	1,5	1
Palmitic Acid	1,5	0.4
Stearic Acid	1	0.5
2,2'-Bipyridine	4	4

*1 - Lake Marion, 2 - Fort Loudon Lake, 3 - Holston River, 4 - Mississippi River, 5 - Watts Bar Lake fate. For instance, acetone is infinitely soluble in water; chloroform is soluble to about 8,200 mg/ℓ; carbon tetrachloride is soluble to about 800 mg/ℓ; and n-decane is insoluble. The specific gravity of toluene is less than that of water, while the specific gravity of carbon disulfide is greater. Acetaldehyde is readily metabolized since it is a natural metabolic intermediate, but branched alkyls are almost impervious to microbial attack.

The differences in man-made synthetic/organic compounds exceed the similarities, but in general, these compounds are persistent and resist microbial degradation. Beyond that generalization, research has been too limited to discuss specific compounds in detail.

Polychlorinated biphenyls (PCB's) are the only class of synthetic/organic contaminants that have been studied in detail. They are virtually insoluble in water, which, combined with a high specific gravity and volatility, serves to keep solution PCB concentrations low. However, PCB's are strongly adsorbed onto suspended particulate matter and transported through the water system. Because of their heavy, insoluble character and sediment affinity, they tend to accumulate in bottom sediments. A comparison of selected water and sediment PCB concentrations from across the United States is presented in Table 80 .

PCB's are fairly stable in fresh-water systems, resisting hydrolysis and chemical degradation, and are not amenable to photodegradation (998). Theoretically, they should readily vaporize from solutions, but this is prevented by their tendency to sink or strongly adsorb onto suspended matter. Only PCB's that are associated with floatables or oil slicks appear to vaporize to any great degree. The lower isomers (four or fewer chlorine atoms) are somewhat responsive to biodegradation, but the degradation products are frequently more toxic than the PCB itself (998). The higher isomers resist microbial attack. PCB's are thus quite persistent in water/sediment systems, and lifetimes of years or even decades have been postulated (998).

The continued presence of PCB's makes it inevitable that they will enter the food chain. As they tend to accumulate in lipid tissues in higher plants and animals, it has been estimated that PCB's will concentrate up the food chain to as much as 10^7 times the water concentration (999).

BIOLOGICAL CONTAMINANTS

An important pathway for certain communicable disease transmission to man is the consumption of contaminated water. Direct disposal of wastewater is the principal contamination route. Land disposal of wastewater and sludge is not an important pathway, as pathogenic organisms have limited mobility

TABLE 80. PCB CONCENTRATIONS IN SELECTED WATER COURSES (275)

State	Concentration Water µg/ℓ	Concentration Sediment µg/kg
Alaska	ND	ND
Arkansas	ND	20-2,400
California	0.1, 0.1	20-190
Connecticut	0.1-0.2	40
Hawaii	ND	ND
Georgia	--	10-1,300
Maryland	0.1	10-1,200
Mississippi	ND	50; 170
New Jersey	0.1	8-250
Oregon	ND	15; 140
Pennsylvania	0.2	10-50
South Carolina	--	30-200
Texas	0.1-3.0	7.9-290
Virginia	0.1	5-80
Washington	ND	ND
West Virginia	ND	10

in soil and seldom migrate far enough to contaminate water supplies (588). However, in contrast to their restricted mobility in soils, biological contaminants are readily dispersed and transported by receiving waters. Consequently, there is a high potential for direct public contact through drinking or recreational use. Wastewater treatment has diminished this threat by reducing the number of organisms in the wastewater. This, combined with natural pathogen mortalities, has greatly lessened the outbreak of water-borne diseases attributable to public water supplies.

The environmental factors that influence the survival of pathogens in fresh water are, in most cases, similar to those that prevail in marine systems. These factors will be discussed in greater detail in the marine water section of this chapter. Briefly, however, the chief factors influencing survivability are temperature, pH, sunlight, toxins, predators, and lack of nutrition, which affect pathogens to different degrees. An examination of Table 81 reveals that pathogens may survive for long periods of time and travel great distances before destruction by environmental factors.

Pathogenic bacteria are best adapted to survival in the human body or to conditions resembling those found in the body. Consequently, natural water systems are a hostile environment. However, cool water is generally more hospitable than warm water because of the depressed metabolism of both the bacteria and their predators. Predatory organisms, especially in slightly polluted waters, are a major contributor to bacterial die-off. For instance, Barua (74) noted that *Vibrio cholerae* survived one to two weeks in clean water as opposed to one to two days in water with a large bacterial population.

Pathogenic bacteria also suffer from a lack of proper nutrition in clean waters; low nutrient levels prevent reproduction. Since die-off rates exceed growth rates, the overall population will decline. Other factors affecting die-off are ultraviolet radiation in sunlight, pH extremes, natural antibiotics, and chemical toxins.

In contrast to bacteria, viruses do not multiply in water and, therefore, their number in a water body can never exceed the number introduced into that body by waste disposal. Typically, viruses are much more resistant to external environmental factors (chemical content, pH, temperature, time, etc.) and survive longer than bacteria (257, 380). It was long suspected that algae could inactivate viruses through some process because of low virus concentrations in algae-rich waters. However, it is now believed that the high pH and dissolved oxygen in the vicinity of algal blooms are responsible for the inactivation.

TABLE 81. AVERAGE TIME IN DAYS FOR 99.9% REDUCTION OF ORIGINAL TITER OF INDICATED MICROORGANISMS IN WATERS (1283)

Microorganism	Little Miami River (Ohio)			Ohio River (Ohio)		
	28°C	20°C	4°C	28°C	20°C	4°C
Polio virus I	17	20	27	11	13	19
Echo 7	12	16	26	5	7	15
Echo 12	5	12	33	3	5	19
Coxsackie A9	8	8	10	5	8	20
A. aerogenes	6	8	15	15	18	44
E. coli	6	7	10	5	5	11
S. fecalis	6	8	17	18	18	57

Virus inactivation in lake water is further enhanced by the presence of proteolytic bacteria which degrade the viral coat (564). Coxsackie is particularly susceptible to proteolytic bacteria, while polio virus is generally resistant except to *Pseudomonas aeruginosa* (564). Otherwise, the mechanisms of virus removal are obscure. Table 82 reports survival times for various enteric viruses in fresh-water bodies.

Transport mechanisms for pathogens include physical current motion, organism motility, adsorption, ingestion, and aerosolization. As most pathogens readily adsorb onto suspended matter, sediment pathogen concentrations may greatly exceed water concentrations. Filter feeding organisms, such as fresh-water shellfish, tend to concentrate pathogenic organisms. Consequently, shellfish can be a major factor in the spreading of certain communicable diseases.

TABLE 82. SURVIVAL OF ENTERIC VIRUSES IN WATER (11)

		Temperature °C		
Type of Water	Virus	4-6	15-16	20-25
		(Range of Days)		
River water	Coxsackie B-3	75/3	8/3	2/3
	ECHO 5	7/0.5	-	3/3
	Polio 1	7/1	-	3/3
	Coxsackie B-3	7/1.7	-	3/3
	ECHO 12	19/3	-	5/3
	ECHO 7	15/3	-	7/3
	Coxsackie A-9	10/3	-	8/3
	Polio 2	75/3	15/3	8/3
	Polio 3	30/3	8/3	8/3
	ECHO 5	60/3	15/3	8/3
	Coxsackie A-9	20/3	-	8/3
	ECHO 12	33/3	-	12/3
	Polio 1	19/3	-	13/3
	Polio 1	60/3	45/3	16/3
	ECHO 7	26/3	-	16/3
	Polio 1	27/3	-	20/3
	Polio 3	50/3	18/3	-
	Polio 3	67/3	7/1.3	7/2.1
	Coxsackie A-2	-	-	5/2
	Coxsackie A-2	-	-	47/2
	Coxsackie B-5	-	24/1	-
	Coxsackie B	18/2	-	-
Impounded fresh water	Coxsackie B-3	7/1.7	-	3/3
	Polio 1	7/1.5	-	3/3
	ECHO 7	22/3	-	4/3
	ECHO 6	5/3	-	5/3
	Coxsackie A-9	6/3	-	6/3
	Polio 1	27/3	-	6/3
	Coxsackie ?	18/3	-	8/3
	ECHO 12	14/3	-	6/3
	Polio ?	21/3	-	10/3
	ECHO ?	23/3	-	12/3
	ECHO ?	21/3	-	20/3
	Polio ?	52/3	-	21/3
	Polio ?	52/3	-	22/3
	ECHO ?	42/3	-	24/3

SALINE WATER

INTRODUCTION

The ocean has traditionally been the depository for a wide variety of civilization's wastes. Previously, the ocean was considered an infinite sink that was not affected by the waters discharged into it. Only recently has man recognized that wastewater effluents and sludges can have an adverse effect upon ocean waters.

Wastewater (treated and untreated) and wastewater sludges are discharged directly to the ocean offshore through submarine outfall pipelines. Beach outfalls are occasionally used for wastewater discharge, although this usually results in contamination of the beach. Sludges are also barged and dumped into the open ocean.

The potential significance of a given discharge varies with the characteristics of the discharge and of the receiving waters. For example, discharges into a bay or estuary are much less diluted than discharges into the open ocean. The location of the outfall (e.g., on a shelf or near a submarine canyon) strongly affects contaminant mobility. Seafood obtained from open ocean areas is different from that harvested from bays or estuaries. Shellfish, which are usually taken from bays or estuaries, tend to concentrate contaminants to a greater degree than does open ocean seafood. Furthermore, the physico-chemical relationships affecting solubility and mass transport of contaminants differ considerably for estuaries and open ocean receiving waters.

Consequently, the fate of wastewater contaminants in marine environments is highly site specific. For any given site, an understanding of the potential pathways from the effluent back to man requires knowledge of the dilution achieved, discharge location hydraulics, chemical and physical composition of both the effluent and receiving waters, composition and hydraulics of other contamination sources, physical and chemical properties of the sediments and their interactions with the water column, and an understanding of the food web relationships. Food web relationships are perhaps the most important element in terms of ultimate pathways to man, and have received the most publicity in recent years.

Public health-impairing contaminants behave in somewhat the same fashion in marine water as in fresh water. Therefore, concepts discussed extensively in the preceding fresh-water section of this chapter will be mentioned only briefly here. In general, wastewater contaminants in the ocean can return to man via two pathways: ingestion of contaminated fish and shellfish, or body contact and incidental ingestion associated with marine aquatic sports.

Research concerning the disposal of sewage effluents and sludges to marine water systems is abundant, as can be seen in Table 83 . The volume of literature is somewhat deceptive, however, in that most of these studies concentrate on one aspect of a pollution problem in a specific area; comprehensive generalizations are usually not possible. The most nearly complete picture of the pathways and fates associated with wastewater discharges comes from a study of the Southern California coastal area by the Southern California Coastal Water Research Project (SCCWRP), which began in 1969. Figure 21 shows the study area, which encompasses approximately 400 m of coast and 20,000 km^2 of water surface area. This coastal area receives municipal wastewater, sludge, surface runoff, and other discharges from a population of approximately 11 million people. The effluent volume from municipal treatment systems alone is approximately one billion gpd, of which 20 percent receives secondary treatment while 80 percent receives only primary treatment (1544).

Because of the completeness of the research, much of the following discussion relies heavily on the SCCWRP data. However, this data is specific to the Southern California area, which differs from other ocean areas; hence, the data must be applied with caution.

WATER QUALITY PARAMETERS

Since saline water is not used for drinking water, there are no potential direct health problems from marine water quality parameters. However, as in fresh surface waters, there are some indirect consequences from the introduction of suspended solids and organic carbon into the marine environment. Trace metals, bacteria, viruses, and other hazardous contaminants are often found in association with suspended particulates. Researchers in Southern California have determined that more than 90 percent of the possible pollutants discharged through one sludge discharge pipeline are associated with particulate matter (925).

Discharged organic matter can have a radical impact on the local environment which, in turn, will affect the distribution, transport, and availability of other contaminants. The oxidation of organic matter, particularly in bottom sediments, severely depletes dissolved oxygen concentrations,

TABLE 83 . LITERATURE REVIEWED
PERTAINING TO SALINE WATER

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	61, 221, 264, 358, 829, 925, 1002, 1102, 1125, 1149, 1207, 1541
BOD	226, 338, 925, 979, 1002, 1102, 1207, 1268
COD	925, 1268
Chlorides	225, 1125
Fluorides	1451
Nitrates	61, 86, 221, 225, 264, 358, 829, 882, 883, 1002, 1125
Nitrites	61, 86, 221, 264, 358, 829, 1002, 1125
Oil and grease	61, 925, 1102, 1207, 1231
Phosphates	61, 86, 221, 264, 358, 829, 882, 883, 925, 1101, 1102, 1207
Suspended solids	459, 499, 925, 1125, 1171, 1172, 1541
Total dissolved solids	1125
Total organic carbon	225, 559, 720, 979, 1125, 1268
Other (general)	42, 226, 551, 552, 700, 828, 925, 1067, 1102, 1125, 1207, 1307, 1310, 1544
Elemental Contaminants	
Aluminum	28, 436, 636, 720, 968, 1125, 1128, 1160, 1161, 1266, 1380, 1413, 1459
Antimony	303, 436, 636, 720, 1125, 1128, 1160, 1161, 1266, 1380, 1525
Arsenic	303, 636, 637, 720, 1128, 1160, 1161, 1207, 1266, 1339, 1380, 1525

TABLE 83. (continued)

Contaminant	Reference Number
Barium	28, 636, 968, 1128, 1160, 1161, 1266, 1339, 1380, 1459
Beryllium	636, 1128, 1161, 1266, 1339, 1380, 1459
Boron	28, 636, 1113, 1128, 1160, 1161, 1266, 1339, 1380, 1459, 1525
Cadmium	53, 75, 115, 157, 227, 303, 365, 441, 524, 558, 592, 636, 713, 720, 756, 828, 923, 925, 967, 968, 978, 1102, 1117, 1121, 1125, 1128, 1160, 1161, 1207, 1208, 1378, 1433, 1459, 1502, 1541, 1544, 1562
Chromium	53, 75, 157, 212, 227, 232, 365, 611, 632, 636, 700, 713, 720, 828, 923, 967, 969, 1102, 1121, 1125, 1128, 1137, 1160, 1207, 1208, 1266, 1306, 1307, 1380, 1433, 1459, 1528, 1541
Cobalt	75, 115, 157, 303, 365, 407, 445, 636, 713, 782, 828, 894, 923, 967, 968, 1044, 1102, 1117, 1121, 1125, 1128, 1160, 1161, 1266, 1339, 1380, 1459, 1538, 1544
Copper	75, 115, 157, 227, 232, 305, 365, 407, 445, 611, 636, 713, 720, 782, 828, 844, 894, 923, 967, 968, 1044, 1066, 1102, 1121, 1125, 1128, 1137, 1159, 1160, 1161, 1171, 1207, 1208, 1259, 1266, 1339, 1380, 1433, 1459, 1538, 1541, 1543, 1544, 1562
Germanium	636, 1128, 1160, 1161, 1266, 1339, 1380
Iron	28, 75, 115, 227, 232, 365, 611, 636, 720, 782, 828, 844, 968, 1113, 1117, 1121, 1125, 1128, 1160, 1266, 1339, 1371, 1380, 1459, 1502, 1544, 1562
Lead	75, 115, 157, 227, 365, 407, 441, 462, 524, 558, 611, 636, 713, 720, 828, 923, 967, 968, 1060, 1066, 1102, 1117, 1121, 1128, 1161, 1171, 1207, 1208, 1266, 1307, 1339, 1363, 1380, 1433, 1459, 1544

TABLE 83. (continued)

Contaminant	Reference Number
Manganese	75, 227, 232, 365, 445, 611, 636, 713, 720, 828, 844, 923, 967, 1102, 1117, 1121, 1125, 1171, 1433, 1459, 1502, 1538
Mercury	12, 37, 227, 303, 305, 365, 368, 452, 462, 524, 610, 620, 636, 637, 645, 654, 760, 780, 795, 809, 967, 968, 987, 988, 1102, 1125, 1128, 1145, 1160, 1161, 1207, 1208, 1266, 1295, 1339, 1380, 1413, 1433, 1502, 1503, 1526, 1527, 1543, 1544, 1562
Molybdenum	303, 636, 720, 828, 925, 1128, 1160, 1161, 1266, 1339, 1380, 1459
Nickel	75, 115, 232, 365, 445, 636, 713, 720, 782, 828, 923, 967, 968, 1102, 1117, 1121, 1125, 1128, 1160, 1161, 1171, 1207, 1208, 1266, 1339, 1380, 1459, 1541
Selenium	636, 1114, 1128, 1160, 1207, 1266, 1339, 1380, 1525
Thorium	636, 1128, 1160, 1161, 1266, 1339, 1380
Tin	636, 828, 923, 1128, 1160, 1161, 1266, 1339, 1380, 1459
Uranium	636, 1128, 1160, 1161, 1266, 1339, 1380
Zinc	75, 115, 157, 175, 227, 232, 303, 305, 365, 407, 441, 445, 524, 592, 611, 636, 713, 720, 782, 828, 844, 923, 968, 1066, 1102, 1114, 1117, 1125, 1128, 1160, 1161, 1171, 1207, 1266, 1307, 1339, 1380, 1433, 1459, 1538, 1562
Other (general)	37, 42, 75, 157, 228, 636, 720, 893, 923, 967, 1161, 1207, 1307, 1339, 1380, 1454, 1460
Biocidal Contaminants	
Aldrin	403, 1237, 1418
Arsenated hydrocarbons	1237

TABLE 83.(continued)

Contaminant	Reference Number
Chlorinated hydrocarbons	304, 324, 365, 605, 804, 827, 874, 893, 977, 998, 999, 1022, 1023, 1059, 1065, 1066, 1068, 1102, 1125, 1132, 1146, 1162, 1170, 1207, 1231, 1237, 1307, 1412, 1532, 1535, 1536, 1539, 1543
DDD	403, 1022, 1125
DDE	403, 977, 1125
DDT	17, 365, 403, 420, 605, 872, 874, 894, 977, 1022, 1023, 1059, 1125, 1170, 1207, 1307, 1418, 1532, 1535, 1536, 1539, 1542, 1543
Dieldrin	365, 605, 1125, 1155, 1237, 1418, 1536
Endrin	403, 1418
Herbicides	806, 1125, 1412
Organophosphorus pesticides	1196
Soil sterilants	73
Other (general)	42, 130, 304, 478, 806, 827, 1049, 1102, 1125, 1407, 1454
Synthetic/Organic Contaminants	42, 58, 149, 235, 300, 304, 318, 319, 365, 402, 653, 768, 770, 807, 808, 879, 893, 903, 1025, 1125, 1126, 1132, 1454, 1533, 1539, 1558, 1564
Biological Contaminants	
Adeno virus	350
Bacteria	18, 44, 45, 74, 365, 569, 656, 674, 858, 929, 938, 987, 1047, 1200, 1202, 1307, 1345
Clostridium welchi	858

TABLE 83.(continued)

Contaminant	Reference Number
Coliforms	17, 18, 44, 61, 197, 253, 254, 304, 310, 311. 316. 338, 612, 613, 656, 674, 675, 677, 695, 705, 938, 973, 989, 1019, 1020, 1021, 1047, 1087, 1200, 1202, 1222, 1223, 1224, 1226, 1231, 1268, 1275, 1307, 1413
Coxsackie virus (A & B)	11, 252, 564, 905
ECHO virus	252, 357, 905, 1515
Escherichia coli	929, 1275, 1306, 1307, 1443, 1515
Fecal streptococci	197, 399, 612, 1202, 1275, 1307
Listeria monocytogens	989
Mycobacterium	1306
Parasitic worms	700
Polio virus	11, 252, 564, 905
Salmonella	1234, 1306, 1307
Shigella	1306
Staphylococcus aureus	350, 1306, 1307
Vibrio cholerae	255, 1020, 1306, 1316
Virus	11, 251, 252, 255, 311, 365, 405, 465, 468, 695, 877, 905, 926, 938, 1273, 1306, 1307, 1316, 1317, 1515
Other (general)	250, 311, 433, 858, 1125

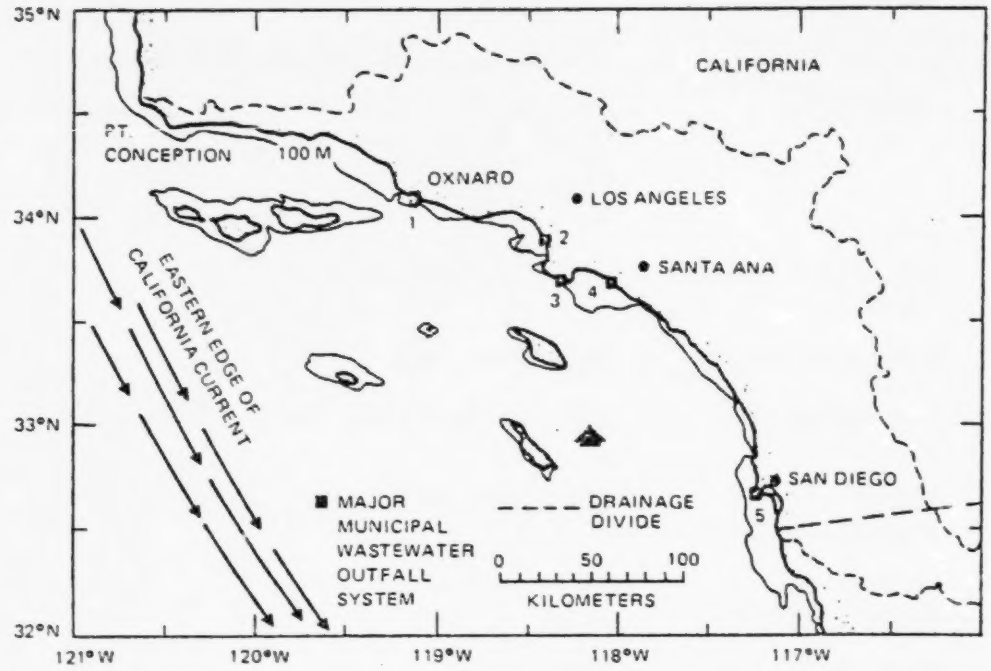

Figure 21. The Southern California Bight. Outfall systems are (1) Oxnard City, (2) Hyperion, Los Angeles City, (3) Whites Point, Los Angeles County, (4) Orange County, and (5) San Diego City. (1544)

leading to anaerobiosis and a reducing environment. In such an environment most dissolved metals will precipitate as sulfides. The presence of dissolved organics enriches the local environment in nutrients and leads to increased planktonic growth and overall biological activity. Fish and crustacean populations may also increase. Since these regions of high organic content may also be high in hazardous elementals and organics, the chances of these fish and crustaceans becoming contaminated are higher than for normal ocean water populations.

Another problem associated with water quality parameters is floatable material. Floatables from wastewater generally can be classified as oil and grease, and methylene blue active substance (MBAS). Again, the hazard is not with these materials but with associated contaminants. Bacteria have been shown to concentrate in floatables (365). Although research has been scanty, it is probable that viruses, heavy metals, biocides, and synthetic organics will also concentrate in floatable materials. These floatables can be carried along the water surface to the shore by wind, current, and wave activity. Once deposited along the beach, these contaminants become readily accessible to both humans and insect/mammal vectors. Wind and wave action can also agitate floatables to the point where aerosol-type particulates are released. These airborne particulates can likewise be high in hazardous contaminants.

As in fresh water, the addition of nitrogen and phosphorus species to marine water can result in biostimulation. Natural seawater concentrations of phosphorus and nitrogen range from 1 to 100 ppb and 10 to 700 ppb, respectively. Discharges of nitrogen and phosphorus to open ocean waters are seldom a problem, as the ocean can assimilate these nutrients. Discharges into bays, however, can lead to accumulations of nitrogen and phosphorus with subsequent biostimulation. Copeland's study of St. Joseph Bay, Florida (264), found phosphorus concentrations as high as 10 times the normal and gross biostimulation as evidenced by massive algal growths. These growths decreased the aesthetic appeal of the affected water and drove away many food and sport fish. Dissolved oxygen levels also decreased, and the bay exhibited evidence of eutrophication.

ELEMENTAL CONTAMINANTS

Marine elemental chemistry is similar in many respects to fresh-water elemental chemistry. The same physical and chemical principles apply, but the marine system is somewhat more complex. Descriptions of the principles of solution, precipitation, adsorption, etc., given in the fresh-water section of this report will not be repeated. Whenever the complexities of the marine system introduce modifications in principles already discussed, these will be mentioned. In general, however, this discussion will be limited to a simple description of some of the important components of the marine system and their interactions, with little mention of the underlying principles.

The concentrations of elemental contaminants entering the marine system via sewage effluent or sludge vary widely depending on the source of the effluent or sludge (Table 84). Recent research has shown that 50 to 90 percent of these elemental contaminants are associated with the particulate fraction of the discharge (558, 967). Analyses of sediments around outfalls seem to indicate that this elemental/sewage particulate association does not last long in seawater, although the mechanism of release is not well understood. Among the possible explanations are:

- Oxidation of element-containing organic particulates with subsequent release of the element
- Oxidation of particulate element sulfides
- Surface desorption caused by high dilution ratios
- Complexation with inorganic ligands such as $C\ell^{-2}$
- Complexation with organic ligands possibly resulting from the oxidation of organic particulates.

The behavior of the elemental contaminants after their release from the sewage particulates is scarcely better understood. Table 85 presents the average natural marine concentrations and principal dissolved species for several of these elements. It should be noted that these principal species were probably identified in open ocean waters; dissolved elementals may take different forms in coastal waters. Furthermore the chemistry of the sewage-rich ocean waters near an outfall can be radically different from the chemistry of the open ocean.

Sewage-borne elemental contaminants are concentrated and generally associated with suspended matter. At sludge outfalls, reducing conditions that prevail may lead to the formation of insoluble sulfide compounds. The organics in the sewage can lead to a reducing or only moderately oxidizing environment where many metals -- such as zinc, lead, cobalt, cadmium, and copper -- form insoluble carbonate compounds. Chromium

TABLE 84 . SEWAGE RELATED INPUTS OF ELEMENTAL CONTAMINANTS TO MARINE SYSTEMS

Elemental 2	Wastewater I (1207) µg/ℓ	Wastewater II (1207) µg/ℓ	Sludge I (1102) mg/dry kg	Sludge II (1459) ng/hg
Arsenic	<11	10	--	--
Barium	--	--	--	2,000
Beryllium	--	--	--	1
Boron	--	--	--	100
Cadmium	36	20	3.0 - 76	100
Chromium	800	130	2.0 - 3,700	4,000
Cobalt	--	--	--	50
Copper	420	190	26.0 - 11,955	4,800
Lead	250	30	5.0 - 1,208	6,000
Manganese	--	--	10.0 - 1,530	4,000
Mercury	1.1	2	0.2 - 85	--
Molybdenum	--	--	--	50
Nickel	280	150	4.0 - 1,100	1,000
Selenium	<13	20	--	--
Tin	--	--	--	1,000
Zinc	1450	230	93.0 - 48,410	20,000

TABLE 85. CONCENTRATION OF SELECTED ELEMENTS IN MARINE WATER (1128)

Element	Concentration ($\mu g/\ell$)	Principal Dissolved Species
Aluminum	1	$Al(OH)_y^{(3-y)} \cdot (H_2O)_n$
Antimony	0.3	
Arsenic	2.6	$HAsO_4^{-2}$, $H_2AsO_4^-$
Barium	20	Ba^{+2}
Beryllium	6×10^{-4}	
Boron	4.5×10^3	$B(OH)_3$, $B(OH)_4^-$
Cadmium	0.1	Cd^{+2}, $CdOH^+$, $CdCl^+$
Chromium	0.5	CrO_4^{-2}, Cr^{+3}
Cobalt	0.4	Co^{+2}
Copper	3	Cu^{+2}, $Cu(OH)_2$
Germanium	0.07	$Ge(OH)_4$
Iron	3	
Lead	0.03	$PbCl_3^-$, $PbCl^+$, Pb^{+2}
Manganese	2	Mn^{+2}
Mercury	0.2	$HgCl_4^{-2}$, $HgCl_2$
Molybdenum	10	MoO_4^{-2}
Nickel	7	Ni^{+2}, $NiOH^+$, $NiCl^+$
Selenium	0.09	SeO_4^{-2}
Thorium	$<5 \times 10^{-4}$	$Th(OH)_4$
Tin	0.8	
Uranium	3	$UO_2(CO_3)_3^{-4}$
Zinc	10	Zn^{+2}

which is predominantly trivalent in sewage (632), forms insoluble hydroxides and oxides.

Morel et al. (967) and Hendricks (558) indicated that less than 10 percent of the particulate elements settle in the immediate outfall area. Rather they are carried away by current action, and the dilution and oxidation potential are increased significantly. When thus spread and diluted, many of the precipitates formed near the outfalls will dissolve, e.g., sulfides will oxidize to soluble sulfates, and carbonate compounds will dissolve due to the action of competing ligands.

Once in solution in open water, dissolved elements are subject to a wide variety of competing mechanisms that determine whether a given elemental remains in solution or is removed. In highly dilute, oxygenated seawater, most elements are in solution as:

- Cations: Ba^{+2}, CO^{+2}, Cu^{+2}, Ni^{+2}
- Oxo anions: $HAsO_4^{-2}$, CrO_4^{-2}, SeO_4^{-2}, MoO_4^{-2}
- Chloride complexes: $CdCl^{+}$, $PbCl_3^{-}$, $HgCl_4^{-2}$, $NiCl^{+}$
- Hydroxyl complexes: $B(OH)_3$, $B(OH)_4$, $Cu(OH)_2$
- Carbonate complexes: $UO_2(CO_3)_3^{-4}$.

Iron and manganese are major exceptions to the solubility rules; both are more soluble in their lower valence states, and as a result, are mobilized in reduced sediments and enter solution as the divalent cations.

When divalent iron and manganese migrate to more highly oxygenated waters, they oxidize to form insoluble hydrous oxides that precipitate. These hydrous oxide particulates have a high affinity for other ions and act somewhat as scavengers. They are particularly effective at adsorbing cobalt, nickel, cadmium, zinc, silver, selenium, and lead but will also remove some chromium, copper, uranium, and molybdenum.

If these hydrous iron and manganese oxides settle into reducing sediments, the iron and manganese are again released into solution. The accompanying ions, however, are usually locked into the sediments as the extremely low solubility carbonate or sulfide compound.

If these hydrous oxides settle into oxidizing sediments, the adsorbed ions generally become incorporated into the precipitate. The iron and manganese oxides accrete to form the ferro-manganese nodules that cover much of the ocean floor. These nodules are invariably enriched in lead, cobalt, nickel, zinc, cadmium, etc. A major exception to this behavior is chromium, which is often carried with the hydrous oxides as the Cr(III) oxide. The redox conditions around the iron/ manganese minerals are such, however, that the Cr(III) is oxidized to the highly soluble Cr(VI), which is then released back into solution. Most of the elementals, once they become incorporated into the sediments, are effectively locked away from further reaction with the seawater, barring biological activity or a major change in redox potential (Table 86).

Biological activity also plays an important role in the transport of elemental contaminants. Biologically related mechanisms take several forms: uptake and biomagnification by organisms, biological transformation, sedimentation associated with dead organisms or fecal matter, and complexes with biologically related organic ligands.

Generally, elemental contaminants become associated with marine organisms via one of several pathways. Some trace metals (e.g., cobalt, iron, copper, zinc) can be incorporated directly into metabolic pathways. Other trace metals (notably mercury) exhibit lipid-preferential solubilities whereby they concentrate in the fatty cells of small organisms, the gill tissues of fish, or the digestive organs of filter feeders. Metals can adsorb onto the surface of plankton or can be transferred to higher marine organisms (and ultimately to terrestrial organisms and man) via ingestion of contaminated food organisms. Two other pathways, bioaccumulation and biomagnification, are discussed in more detail in the "Pathways to Man" section of this report, under "Fish" and "Shellfish."

Incorporated elemental contaminants can be returned to the sediments in fecal material or in dead organisms. If this occurs in a biologically active area, the elementals can be ingested or sorbed by other organisms or simply released again to solution; otherwise they will probably be incorporated into the sediments. Sediment elementals (whether there by chemical or biological means) are subject to ingestion by bottom feeders or transformation by sediment bacteria. Biotransformation is probably an important transport mechanism. For example, inorganic mercury and its compounds, chemically immobile in sediments, are readily absorbed by many bacteria. These bacteria, possibly via a detoxification mechanism, convert the inorganic mercury to a organomercurial. The organomercurial, usually methyl mercury, is far more mobile than its inorganic counterpart. It is

TABLE 86 . NATURAL TRACE METAL CONCENTRATIONS (MG/DRY KG) AS REPORTED FOR SURFACE SEDIMENTS FROM SEVERAL PARTS OF THE WORLD OCEAN (365)

Trace Metal	So. Calif.[a]	Other Nearshore	Organic Rich Cont. Shelf[b]	Pacific Basins[c]	Manganese Nodules[d]	Outfall Max.[e]
Silver	1.0					21
Cadmium	0.4	2[f]				79
Cobalt	7	5[g]		4-195	3,000	11
Chromium	46	30[g]		4-93		1,000
Copper	16	7[h], 48[i]	18-129	82-686	5,000	670
Iron	2.5%	1-5%[j]		0.4-5.5%	14%	3.7%
Mercury	0.06	0.04[k]		0.09-0.86		4
Manganese	320	580[f]		42-1,075	19%	380
Nickel	14	15[h], 55[i]	35-455		4,000	45
Lead	8	15[h], 20[i]	3-32		1,000	490
Zinc	63	95[i]	18-337		400	2,400

[a] Averages of natural values at the five major sewage outfalls in the Bight. However, mercury value is estimated natural subsurface concentration in Santa Barbara Basin.

[b] Organic-rich diatomaceous muds, S.W. Africa shelf.

[c] Sample No. 21-29, Pacific Ocean Basins.

[d] Average composition of ferromanganese minerals from the Pacific Ocean.

[e] Maximum surface sediment concentration measured off Palos Verdes Peninsula sewage outfalls.

[f] Caribbean sediment, Station 50.

[g] Japanese Island sediments,

[h] Gulf of Paria, Caribbean sediments.

[i] Atlantic nearshore sediments.

[j] Gulf of Paria, Caribbean sediments.

[k] Washington Shelf, N.E. Pacific.

easily released into the water column and is even more lipid preferential than inorganic mercury. Only mercury sulfide compounds and mercury bound to organic sulfhydryl groups are relatively unaffected by bacteria. This type of behavior has also been postulated, or demonstrated in the laboratory, for arsenic, selenium, antimony, lead, cadmium, and other elementals.

Organic ligands from decaying organisms or excretory products also may play a role in the solubility of many elements. Examples are the copper-heme, iron-heme, and cobalt-vitamin B_{12} complexes. All are natural complexes and are extremely stable in marine environments. Several metals, such as mercury and nickel, show a high affinity for organic ligands and particulates. These element-organic ligand complexes will stay in solution until the organic ligand decays, the complex is ingested or sorbed, or the elements can form a more stable compound.

The ocean as a whole is in a relatively steady state or equilibrium condition, i.e., the rate of input of natural materials roughly equals the rate of sedimentation. Consequently, the concentrations of most elemental contaminants in seawater are constant. Generally speaking, those elements present in highest concentrations are the least reactive and least likely to sediment. The converse is equally true. Moreover, the elements considered hazardous to man are among those present at extremely low concentrations. Therefore, it is difficult to see how man's additions can have a serious impact on the levels of these contaminants in seawater. Of course, man has been adding these metals to seawater for decades; but, as was just mentioned, these metals are among the most reactive and most likely to precipitate. The cations, anions, and ligands most responsible for this precipitation are present in excess and are much more readily replenished than are the more hazardous elements. Consequently man's additions can be removed relatively rapidly without seriously affecting marine equilibrium.

While this may sound comforting, it does not take into account the fact that sludge and wastewater disposal occurs in coastal waters where exchange with open ocean waters may be limited. Ocean waters form an open, steady-state system; coastal waters form a partially closed system that is subject to varying inputs from the land. Thus coastal waters are the most biologically active of the ocean's areas; consequently, they are far more readily upset by wastewater additions than is the ocean as a whole.

It would be extremely useful to be able to describe the relative impact different wastewater element concentrations would have on coastal equilibrium models. Unfortunately, as discussed earlier in this section, there is a distinct lack of uniformity among various coastal regions that makes the task virtually impossible. Even the SCCWRP (the most complete characterization of a specific region) does not adequately address the problem of the diverse effects of different wastewater levels. Suffice it to say that, in general, wastewater disposal in coastal waters will increase the concentrations of elements in the water, sediments, and marine organisms.

BIOCIDAL CONTAMINANTS

The behavior of the biocidal contaminants in marine water systems is largely the same as in fresh-water systems. The mechanisms of microbial and chemical degradation, photodegradation, sediment and particulate adsorption, volatilization, and biological uptake are physically the same. The most important difference is the reduced solubility of many of the biocides in the more highly ionic seawater. This will lead either to increased sedimentation or increased concentration in surface films, depending on the density of the particular biocide.

Biocides in marine systems have not been studied as extensively as biocides in fresh-water systems. Most of the research on the former is concerned with uptake by or effects on aquatic life. Little has been done to characterize the physico-chemical transport and removal mechanisms active in seawater. Furthermore, what research has been conducted has been limited largely to DDT.

Table 87 lists the reported concentrations of DDT, PCB, and dieldrin in selected effluents and sludges discharged to the ocean off Southern California. These effluents contain both domestic and industrial sewage and, in some cases, are not treated beyond advanced primary treatment.

The chlorinated hydrocarbons are not metabolized to any extent in seawater, polluted estuaries, or bottom sediments (1059). Rather the bottom sediments, particularly those near the outfalls, act as a sink where these biocides may persist for years and even decades (874).

Suspended or dissolved chlorinated hydrocarbons are subject to uptake by algae. A number of researchers have determined that algae are extremely efficient accumulators of these biocides and their residues (804, 1155, 1418). The algae do not degrade the biocides but store them. The biocides can be released if the algae are placed in clean seawater. Otherwise, the algae retain the biocides until the

TABLE 87. CONCENTRATION OF CHLORINATED HYDROCARBONS (μg/ℓ) IN 1-WEEK COMPOSITE SAMPLES OF FINAL EFFLUENTS FROM MUNICIPAL WASTEWATER DISCHARGERS, SPRING 1972 (365)

	Laboratory A*			Laboratory B**			
Discharger	Total DDT	Total PCB	Dieldrin	Total DDT	Total PCB	Dieldrin	Period Sampled
W9 Oxnard	0.16	0.21	0.01	0.06	0.20	0.04	22-29 May
W10 Hyperion							
Effluent	0.21	0.28	0.03	0.13	0.30	0.06	1-7 May
Sludge	5.0	280	1.3	2.4	63	1.9	1-7 May
W11 JWPCP	9.6	24	0.14	12	3.6	0.12	31 Mar-7 Apr
W16 Orange Co. San. Dist.	0.36	27	0.03	0.36	6.6	0.18	21-27 May
W24 Pt. Loma	0.26	0.92	0.02	0.12	0.92	0.02	7-13 Mar

* University of California, Bodega Bay Institute of Pollution Ecology

** San Jose Creek Pesticide Laboratory of Los Angeles County Sanitation Districts algae are either eaten or die. in which case the biocides will sediment with the decaying organic matter.

Chlorinated hydrocarbons in surface films are subject to photodegradation, but even this may be more of a problem than a solution. For instance, Rogers and Landreth (1170) demonstrated that DDT could photodegrade to PCB's.

Although the sediments act as a sink for many biocides, they do not permanently remove them from the environment. On the contrary, contaminated sediments will act as a source for the biocides long after major reductions have been made in the dominant inputs. Bottom-feeding fish placed in an otherwise clean environment (clean seawater with no biocide inputs but with contaminated sediments) soon show significant increases in tissue biocide content (1539). This problem is limited largely to PCB's and the persistent chlorinated hydrocarbons. Carbamates, organophosphates, and ionic biocides all hydrolize rapidly in seawater, with half-lives of hours or days, seldom exceeding a few weeks.

SYNTHETIC/ORGANIC CONTAMINANTS

Most of the research conducted to date on organic contaminants in the marine environment has been concerned with PCB's and the petroleum hydrocarbons associated with oil spills. Little has been done to determine the environmental fate of the plethora of other synthetic organic compounds released to the ocean. Because of the nonpotability of seawater, little concern has been expressed regarding the fate of the synthetic/organic compounds therein. The lipid solubility of some of these compounds, however, suggests that some research is in order, though the sheer number of compounds may discourage meaningful research.

Polychlorinated biphenyls are one group of synthetic/organic contaminants that has been studied in depth. PCB's are highly insoluble in seawater, adsorb rapidly and strongly onto suspended matter, and concentrate in surface oil slicks or sediment. Adsorbed PCB's will ultimately diffuse into the sediments, while those PCB's concentrated on the surface photodegrade to lower isomers that hydrolize relatively easily (1132).

The research on petroleum hydrocarbons is only partially useful. Municipal wastewater ranks low among marine petroleum hydrocarbon sources; natural seeps, oil spills, aerial fallout, industrial discharge, and deliberate dumping all contribute more petroleum hydrocarbons to the sea. However, as the behavior of many organic compounds in seawater is similar, theoretical pathways can be suggested.

In general, any organic compound that can occur naturally, can be metabolized and effectively removed from solution. Totally synthetic compounds may or may not be metabolized, but those structurally similar to natural compounds are more amenable to microbial attack than those with no natural analogs. Increasing substitution, particularly with halogens, increases resistance to biodegradation.

Most organic compounds are degradable via photolytic oxidation or chemical attack. In some cases, e.g., plastics or chlorinated hydrocarbons, half-lives of years are not unusual. Most organic compounds, however, degrade more quickly; half-lives of hours to perhaps a few weeks are more common.

Chemical degradation may take the form of hydrolysis or reaction with another chemical species in the water. Hydrolysis is a reaction with water and usually consists of adding a hydroxyl group or breaking an ester-type linkage. The number of chemical reactions possible are almost limitless. Furthermore, there is no guarantee that reaction or hydrolysis products will be any less harmful or more biodegradable than the parent compound. For instance, azo dyes are readily attacked by hydrogen sulfide to give carcinogenic, non-biodegradable benzidines (1126).

Photodegradation is limited to surface films, as ultraviolet radiation will not penetrate beyond a few centimeters. Consequently those organic compounds dissolved in surface films, adsorbed onto floatables, or less dense than water are most amenable to photolysis.

Again, in general, municipal effluents are not responsible for most organic contaminants in the oceans. Furthermore, most organics discharged in such effluents, if not degraded, remain in the general vicinity of the outfall. Also, the wide variety and small quantities of most organic contaminants make it unlikely that any one will become a hazard in recreational waters or food organisms. Unfortunately this does not take into account possible synergistic effects of contaminant combinations nor the postulated zero threshhold value for some carcinogens.

BIOLOGICAL CONTAMINANTS

Both untreated and partially treated wastewater and sludge are carriers of a variety of pathogenic organisms. Furthermore, since sewage destined for ocean disposal is often given minimal treatment, pathogen concentrations in effluents and sludges can be extremely high (44).

There are five principal types of human pathogens, and many of the disease-producing organisms therein can be transmitted via water routes. The five types and some of the more prevalent waterborne diseases are:

- Protozoa (amebiasis, balantidiasis)
- Nematodes (angiostrongyliasis)
- Platyhelminths (clonorchiasis)
- Viruses (viral hepatitis, viral meningitis, viral respiratory diseases)
- Bacteria (coliform diarrhea, cholera, leptospirosis, salmonellosis, shigellosis, typhoid fever)

There has been little research on the behavior of pathogenic protozoa, nematodes, and platyhelminths in marine systems. They are not as prevalent in industrialized societies nor generally as serious from a public health standpoint. Furthermore, they are largely host specific with few marine hosts and usually a problem only in fresh-water systems.

Bacteria and viruses are much better represented in the literature. Both are virtually ubiquitous, and the diseases they cause are more prevalent with more serious outbreaks. A number of researchers have attempted to assess the fate of these human pathogens in the marine environment. Unfortunately, many of the studies have been done in the laboratory with simulated seawater, and the results may not always be applicable to real systems. At best, these laboratory studies can only evaluate one or two factors, so the effects of factor combinations and synergisms must remain largely a mystery. Consequently, those factors primarily responsible for pathogen die-off or transport in the sea have not been isolated definitively. Table 88 presents reported marine survival times for viruses and selected bacteria.

The greatest proportion of pathogenic bacteria in municipal effluent is associated with particulate matter (44). As a result, ordinary sedimentation accounts for the initial removal of many bacteria. However, some bottom-feeding fish and shellfish can accumulate bacteria (e.g., *Clostridium perfringens*) and provide favorable living conditions (858). Consumption of raw or partially cooked seafood taken near outfalls then becomes a problem.

Bacterial populations can increase near outfalls as well. The relatively higher nutrient content of the water near an outfall provides a much better growth media than does normal seawater, with a resulting increase in bacterial

TABLE 88 . BIOLOGICAL SURVIVAL TIMES (DAYS)
IN MARINE ENVIRONMENTS

		Source
Viruses	2-130	468
Viruses	<28-130	905
Viruses	15-88	11
Salmonella	>25	1307
	58-69	1234
E. Coli	>25	1307
	> 4	929
	75	1306

populations (1200, 1515). In general, though, normal seawater is considered bacteriostatic, and die-offs rather than population increases prevail in most of the ocean. These die-offs are attributable to a variety of factors, including:

- Sensitivity of many bacteria to oceanic extremes of pH and temperature (365, 674);
- Severe disruption of normal osmotic cell balance due to marine salinities (365);
- Bacteriostatic effects of solar radiation on any organisms near the surface (365);
- Bacterial predators flourishing near outfalls (365, 929);
- Secretion by marine plankton of a fairly potent antibiotic (44, 45); and
- Normal seawater nutrient levels that are too low to support bacterial growth (345, 1200, 1515).

Consequently, bacterial die-off is generally rapid away from the influence of the outfall, except for those bacteria that may be ingested by fish or shellfish.

Bacterial populations will stay high in areas where effluent discharges are continuous, particularly if there is little current mixing. As a result, many bays and estuaries can exhibit pathogen concentrations far beyond normal expectations, and recreational use of the water may become hazardous.

Pathogenic bacteria tend to concentrate in surface films and floatables. Aubert et al. (44) found concurrent coliform concentrations of <1/100 mℓ in the water, 3 to 300/100 mℓ in surface films, and 3,500 to 21,000/particulate in floatables. This raises the possibilities of aerosolization of bacteria or surface transport of bacteria to recreational beach areas.

Viruses are far more resistant to seawater than are bacteria. The relative time required for 90 percent inactivation of viruses in seawater has been estimated to be three to six times that for coliform bacteria. Water temperature seems to be the most important factor in virus inactivation; water temperatures of 25°C and lower yield survival times in months (252). Studies in warm waters, such as off Tel Aviv, have shown 90 percent inactivation in less than 48 hr (365). Die-off times are also largely virus specific, with several studies showing that coxsackie survives best, followed by ECHO virus and then polio virus (252, 905).

Although the viruses survive longer than do bacteria, they do not have the tendency to increase in population as do bacteria (1515). Viruses require living hosts to replicate and are very host specific. Human viruses generally do not reproduce in nonhuman hosts.

Viruses can, however, accumulate in marine organisms. Fish and shellfish can ingest viruses and essentially store them. Viruses survive quite well in a variety of shellfish without harming their hosts (365, 905). Viruses can also concentrate on particulates and floatables with the same consequences as for bacteria.

LAND/GROUNDWATER

INTRODUCTION

Wastewater treatment sludges have traditionally been disposed to land, and wastewater effluents are also increasingly being disposed to land. Land disposal of sludges and effluents, if improperly managed, can result in a potential public health hazard. This hazard can be brought about directly through surface contamination of food crops, or indirectly through contamination of groundwater supplies, consumption of livestock fed with contaminated crops, and surface runoff affecting water supplies, aquatic life, or bodies of water used for recreational activities. Aerosols from spray application of effluents and/or direct vaporization of volatile contaminants may also pose a direct or indirect threat to human health. The major potential problem is the contamination of groundwater resources, which supply a significant percentage of domestic water consumption in the United States.

Many variables affect the potential for a public health hazard from land disposal. These include:

- The characteristics of the sludge or wastewater disposed
- The rate of waste application
- The hydrogeological characteristics of the disposal site
- The method of disposal, e.g., crop irrigation, land spreading, percolation ponds, sanitary landfill, etc.
- Proximity of public access
- Utilization of groundwater, crops, etc., potentially affected
- Local climate.

The above listed variables are not mutually exclusive; they interact in many ways to influence the rate and extent of transport of contaminants from each source and to influence the importance of each potential pathway. It is beyond the scope of this report to delve deeply into the science of waste land application technology. This subject is being researched heavily by the EPA and other agencies to establish guidelines for the safe land disposal of effluents and sludges. This report will, however, discuss the current knowledge about potential public health problems.

Literature reviewed concerning wastewater and sludge disposal to land (tabulated in Table 89) has centered on groundwater elemental and biological contaminants, and water quality parameters providing plant nutrients, such as nitrogen and phosphorus forms. The increased use of wastewater application to land has created a need for increased research into the dynamics of wastewater contaminants in soil systems. It is necessary to study the capabilities of these systems to destroy, neutralize, remove, concentrate, or otherwise affect applied wastewater contaminants.

A number of factors determine the degree to which groundwater may be contaminated by wastewater or sludge that is applied to land. Depth to the groundwater table and distance to an extraction point affect residual levels of phosphorus, bacteria, and other constituents for which removal appears to be a function of travel distance. Soil characteristics, native groundwater quality, assimilation capacity of the aquifer, and method of waste application also determine groundwater degradation and consequent health problems (1214). Cation exchange and adsorptive capacities important in the removal of metal ions and viruses, and of trace organics and solids, respectively are determined by soil composition. Porosity regulates infiltration rates to some extent, affecting contaminant residence time in surface layers. Residence time may, in turn, determine aerobic or anaerobic conditions.

Total groundwater volume cannot necessarily be considered an effective diluting agent. Uniform diffusion of recharged water cannot be guaranteed, and water quality may vary considerably both in area and in depth.

WATER QUALITY PARAMETERS

Research in this area has been principally concerned with nitrogen and phosphorus forms entering groundwater as a result of land application of wastewater and sludge. Removal of suspended solids from wastewater effluent has also received attention.

The problems and transformations associated with nitrogen forms in soils are relatively well known. Organic nitrogen and ammonia, when applied to soils under normal aerobic conditions, are rapidly converted by nitrifying bacteria to nitrite and nitrate. Both of these anionic forms move through soils in

TABLE 89. LITERATURE REVIEWED PERTAINING TO LAND/GROUNDWATER

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	26, 47, 134, 218, 377, 479, 704, 705, 755, 1118, 1214, 1323, 1328, 1484
BOD	26, 134, 1140, 1215, 1233
COD	26, 134, 1148, 1214, 1484
Chlorides	26, 47, 352, 1018, 1215, 1419, 1440, 1484
Fluorides	134, 1119, 1173
Nitrates	26, 47, 134, 218, 272, 377, 479, 690, 704, 705, 776, 798, 1118, 1119, 1167, 1214, 1233, 1323, 1328, 1419, 1440, 1484
Nitrites	47, 218, 479, 705, 798, 1118, 1119, 1284, 1432, 1484
Phosphates	47, 68, 134, 352, 595, 690, 755, 1214, 1299, 1419, 1484
Suspended solids	47, 84, 134, 352, 415, 1140, 1214, 1215, 1233, 1388
Total dissolved solids	26, 134, 352, 1215, 1304, 1323
Total organic carbon	134
Other (general)	106, 169, 393, 916
Elemental Contaminants	
Aluminum	378, 634, 776, 1108, 1323
Arsenic	1323, 1419
Boron	134, 378, 736, 797, 1007, 1024, 1327, 1484
Cadmium	134, 401, 690, 776, 1323

TABLE 89 (continued)

Contaminant	Reference Number
Chromium	776, 1419
Cobalt	634, 776
Copper	134, 378, 401, 634, 690, 736, 776, 797, 1007, 1024, 1323, 1327, 1495
Iron	26, 47, 634, 736, 776, 782, 797, 939, 1007, 1024, 1214, 1285, 1323, 1327
Lead	39, 134, 634, 776, 782, 1323
Manganese	378, 634, 736, 776, 782, 797, 1007, 1024, 1323, 1327
Mercury	128, 134, 401
Nickel	396, 401, 634, 690, 798, 1141, 1423
Selenium	1323
Zinc	134, 378, 634, 690, 736, 776, 797, 1007, 1024, 1323, 1327
Other (general)	634, 661, 690, 1173, 1284, 1440
Biocidal Contaminants	
Chlorinated hydrocarbons	401
DDT	401
Dieldrin	401
Herbicides	1014, 1417, 1474
Other (general)	1173, 1323, 1326
Synthetic/Organic Contaminants	934, 1384
Biological Contaminants	
Bacteria	428, 619, 860, 892, 1105, 1175, 1210, 1304, 1323, 1388, 1419

TABLE 89. (continued)

Contaminant	Reference Number
Coliforms	47, 134, 155, 401, 776, 917, 1129, 1140, 1175, 1215, 1233, 1323, 1440, 1484
Coxsackie virus (A&B)	322
Fecal streptococci	1210
Parasitic worms	1323
Polio virus	322, 774
Protozoa	428, 1304, 1323
Salmonella	401, 1083, 1323, 1440
Virus	134, 322, 352, 428, 892, 1083, 1175, 1210, 1215, 1304, 1323, 1479, 1548
Other (general)	619, 776, 917

percolating water with little difficulty. Under anaerobic soil conditions, on the other hand, the process of conversion to nitrite and nitrate is inhibited. Ammonium ions and free ammonia persist and are held near the soil surface by adsorption onto soil particles, by cation exchange reactions, or by fixation in clay lattices.

In an acidic environment, nitrite has been found to react with secondary amines to produce nitrosamines. These compounds have recently been labeled carcinogenic, teratogenic, and mutagenic. The health hazards associated with nitrite and other forms of nitrogen in drinking water and crops have been delineated by the U.S. Department of Agriculture (1432).

The most definitive study of nitrogen removal by land application of wastewater effluent was conducted by Bouwer, et al. at Flushing Meadows, Arizona (134). They found that short flooding periods (two days flooding followed by five days drying) did not provide sufficient time to develop the anaerobic conditions for nitrate denitrification. A longer flooding period of ten days followed by two weeks of drying proved to be more favorable.

With this schedule, oxygen in the soil was depleted during flooding, causing nitrogen (in the ammonium form) to be adsorbed by the clay and organic particles. Flooding was stopped before the cation exchange complex in the soil was saturated with ammonium. Upon drying, oxygen entered the soil, and ammonium was nitrified under aerobic conditions to nitrate. Concurrently, some of the nitrate formed was denitrified - in micro-anaerobic pockets in the otherwise aerobic upper soil zone - to nitrogen gas that escaped to the atmosphere. When flooding was resumed, if the basins were immediately flooded to a depth above 1 ft, the nitrates were quickly leached out of the top few feet of the soil to groundwater. However, if initial flooding was shallow (a few inches deep), the lower head allowed a low infiltration rate, a larger nitrate retention time in the microbiologically active soil zone, and further denitrification. At these lower initial hydraulic loading rates, nitrogen removals were as high as 80 percent. If high application rates were consistently maintained, nitrogen removal was only 30 percent with a peak nitrate surge to the groundwater after the start of each new flooding cycle.

A study by Preul (1119) in 1966 provided the following observations of the movement and conversion of nitrogen in soil and the potential dangers of nitrate contamination:

1. Biological oxidation is the dominant mechanism affecting ammonia nitrogen as it passes through the soil. This action initially occurs at a high rate and to a large extent within several feet of the point of release of the septic tank effluent, if soil conditions are well aerated.

2. Nitrate contamination of groundwaters is a serious threat from shallow soil adsorption systems. High concentrations of ammonia nitrogen in septic tank effluents are quickly nitrified to high concentrations of nitrate, which pollute the groundwater. Dilution from groundwater or soil moisture and possibly denitrification aid in the deterrence of nitrate.

3. The effectiveness of adsorption in deterring the travel of nitrogen is limited because of the rapid conversion of ammonia to nitrate. Laboratory experiments have shown that ammonium can be readily removed in soil by adsorption but, under aerated soil circumstances, nitrification of these ions occurs before the flow can contact a sufficiently effective volume of soil.

Similarly, results of a study by Chapman et al. (218) have shown that, in Texas, irrigation with a sewage effluent was a potential source of nitrate pollution of the local groundwater. The results indicated that nitrification of ammonia nitrogen in the effluent is rapid and complete, taking place within the top 3 ft of soil. It was concluded that substantial amounts of nitrate would not be fixed by the soil and that, at a 3-in per week application rate, appreciable amounts of high nitrate water would percolate to the groundwater. Selective crop production of grains and grasses having high nitrogen uptakes (corn, bermuda grass, oats) was deemed the most effective method of protecting the groundwater.

Short daily flooding schedules evidently result in highly efficient nitrification to nitrate. McMichael and McKee (892) reported data from test basins in the Whittier Narrows and the Rio Hondo spreading grounds near Los Angeles, California. These basins, equipped to collect water at 2-, 4-, 6-, and 8-ft depths, received treated wastewaters on a daily basis using a short period of flooding and a longer period of drying. This cycle ensured completely aerobic conditions. Studies indicated that almost all the nitrogen had been converted to the nitrate form at depths of 8 ft.

Intermittent and continuous spreading of secondary effluent at the aforementioned Hyperion Treatment Plant (1233) resulted in the nitrogen transformations shown in Table 90 .

TABLE 90 . NITROGEN TRANSFORMATIONS RESULTING FROM DIFFERENT SPREADING TECHNIQUES (1233)

I. Under continuous spreading:	Organic-N	Ammonia-N	Nitrate-N
Effluent applied (mg/ℓ)	7.1-8.5	15.5-17.5	0.2- 0.8
Percolate from 7-10 ft below ground (mg/ℓ)	1.2-2.7	8.7-18.5	5.2-18.1
II. Under intermittent spreading:	Organic-N	Ammonia-N	Nitrate-N
Effluent applied (mg/ℓ)	2.3-3.6	4.9-27.9	0.1-14.2
Percolate from 7-10 ft below ground (mg/ℓ)	1.1-2.1	0.0- 0.7	8.4-22.4

As can be seen from this table, intermittent spreading techniques maintained aerobic conditions in the soil, making possible the oxidation of ammonia and organic nitrogen to nitrate. Under continuous spreading, anaerobic conditions prevailed, and ammonia was still present in significant concentrations at a 7- to 10-ft depth.

When a low-rate application system is used, the amount of nitrogen applied to soil with sewage effluent is not much more than can be removed by crops, according to Bouwer (133). A low-rate system involving application to wheat at 2.5 and 5 cm/week was cited. At the lower application rate, 92 percent of the nitrogen in the wastewater was removed in the soil, while at the 5 cm/week rate, 60 percent was removed. On the other hand, when animal waste slurries or other wastewater with a relatively high nitrogen content are applied, the amount of nitrogen supplied may far exceed that which can be utilized by crops. To obtain significant nitrogen removal under these circumstances, Bouwer suggests that the system be designed to stimulate denitrification in the soil. He cites an instance where this was done by installing an artificial barrier to water movement at a depth of 2 m, causing the formation of an anaerobic region. Ammonia and organic nitrogen in applied wastewater were converted to nitrate in the upper, aerobic region of the soil, which was then denitrified in the lower, anaerobic zone. The system removed 96 to 99 percent of the total nitrogen applied at rates of 1 to 2 cm/day.

A study (1233) conducted at the sewage farm in Arroyo Grande, California, indicated the pulse like effect of intermittent spreading on the nitrate level in the percolate. During weekly flooding of the field with settled sewage effluent, the upper foot of the soil adsorbed 2,840 lb of organic and ammonia nitrogen/ac. As the fields drained and were exposed to oxygen, this adsorbed nitrogen was rapidly converted to nitrate, resulting in nitrate levels of 1,000 to 2,000 mg/ℓ in the soil solution of the top foot. Consequently, the subsurface drainage water exhibited pulses of high nitrate concentration when flooding began. After all the nitrate was leached from the topsoil by the applied wastewater, the nitrate content of the drainage water returned to a low level.

When the South Tahoe Public Utilities Department sprayed treated sewage on forested hillsides in the fall of 1963, nitrogen removals by the soil mantle were more than 65 percent. The removals dropped to 26 percent in the winter when the ground was frozen. Significant amounts of ammonium ion were present in the upper 4 in of soil, but nitrate levels were low at all depths. The removal of nitrogen was attributed to the denitrification in the soil mantle under anaerobic conditions (1233).

The California State Health Department (1233) reviewed nitrogen removals and management at a number of well-injection waste disposal/recharge systems. Results of a six-month study on the injection of slow sand filter effluent are shown in Table 91 .

TABLE 91 . NITROGEN TRANSFORMATION IN RECHARGE AQUIFER, MG/ℓ (1233)

	Organic-N	Ammonia-N	Nitrite-N	Nitrate-N
Natural water in the aquifer	.4-.5	.2	0.0	.4-4.8
Filter effluent before injection	2.2	1.5	.01	21.3
Recharged water in aquifer, 20 ft from injection well	1.4	1.2	.21	18.2
Recharged water mixture in aquifer, 500 ft from injection well	.9	.8	.003	6.1

Analysis of data in Table 91 indicated that the decay of nitrate in the aquifer was due to some extent to denitrification, as well as dilution. Considering such decay, the nitrate nitrogen level was expected to fall to 10 mg/ℓ before traveling a distance of 500 ft in the aquifer. Further tests revealed an anaerobic, microbiologically active zone in the aquifer in the vicinity of the injection well. During the tests, nitrate was largely removed by microbial denitrification within 150 ft of the injection well.

In summary, data indicate that, under proper management conditions, land application of wastewater effluent offers the potential to efficiently remove nitrogen from wastewater and protect groundwater from nitrate contamination. The most successful programs stressed an appropriate flooding/drying schedule to promote both aerobic nitrification and anaerobic denitrification processes, in order to ultimately convert ammonia nitrogen in the wastewater to nitrogen gas. However, if not properly managed, a definite danger exists of polluting groundwater resources with excess nitrates.

A number of investigators have mentioned the possibility of groundwater contamination by nitrogen forms from land application of sludges (377, 690, 705). Yet few quantitative studies were found. Walker (1440) concluded that trenching was a viable method to dispose of sludge with minimal groundwater contamination by nitrates. Results of his study showed no increase in pollutants (with the exception of chlorides) in groundwater monitoring wells for up to 19 months after sludge entrenchment. There was evidence that nitrate pollution would become a problem when sludge dried out and became aerobic. In order to prevent excessive nitrogen from reaching groundwater, it was recommended that the site be underdrained, and that the drained water be used to irrigate surrounding cropland.

Five sites receiving applications of municipal sewage sludge were examined by Prothero (1124). At all sites, the ammonia nitrogen and total nitrogen concentrations of the treated soils compared closely to the control soils at a depth of 50 cm in the soil profile. The nitrate concentrations at this depth were above those of the control soils and increased with increasing application rate.

Brown (151) discussed sludge application to croplands. He stated that serious nitrate pollution becomes a potential problem only when the plant-available nitrogen application to soils exceeds the sum of gaseous nitrogen losses (through denitrification and volatilization) and the nitrogen requirements of the crop. He divided the nitrogen content of sludge into the soluble ammonia fraction (which is immediately available for plant uptake) and the organic nitrogen fraction (which must be converted to inorganic nitrogen by soil microorganisms before it is available). The conversion of organic nitrogen is estimated to occur at a rate of 10 percent the first year that the sludge is applied and 5 percent per year in the following years. The 90 percent of organic nitrogen that is not converted the first year is incorporated into humus-like substances. As this organic nitrogen accumulates in the soil, the rate of sludge application must be decreased if the balance is to be obtained between yearly inputs and losses of plant-available nitrogen to and from the soil.

Brown cited a study which shows that the nitrate content of drainage waters from a sludge disposal site will be excessively high if no restrictions are placed on the quantity of nitrogenous material applied to the soil. When 10 in/yr of digested sludge were applied to soil lysimeters, the drainage waters contained up to 449 mg/ℓ N.

Since 1971, the Metropolitan Sanitary District of Greater Chicago has been monitoring a digested sludge land application site. A 1974 report (1556) states that increases in nitrite and nitrate concentrations have not occurred in groundwater observation wells in areas receiving the sludge, indicating that these constituents are not migrating into the aquifer.

Recent studies of land application of wastewater effluent indicate that the soil system is highly efficient in removing phosphates from wastewater. Phosphate removal is both a function of soil composition and travel distance. In most soils, phosphorus not taken up by plants is immobilized due to the adsorption of phosphate onto the soil. Adsorption is followed by fixation by iron and aluminum oxides if the soil is acid, or by precipitation into various forms of calcium phosphate if the soil is basic (133). These reaction products are sufficiently insoluble, so that phosphorus is held in the upper few centimeters of most soils, and very little phosphorus moves into the groundwater (798). However, in the case of acidic, sandy soils with no iron or aluminum oxides, little phosphate is fixed. Thus, it may be necessary to remove phosphorus from wastewater before its application to such soils (133).

Hook et al. (595) reported that under proper management, most of the phosphorus in wastewater remains in the soil at the disposal site or leaves as a nutrient in harvested crops. They found that soils differed in their abilities to retain phosphorus. In a heavy-textured soil high in iron and aluminum oxides and hydroxides (sesquioxides), phosphorus from effluent irrigation did not increase in the soil below a depth of 1 ft after 7 yr of irrigation. In a light-textured soil with half as much sesquioxides, phosphorus content increased to a depth of 3 ft after 6 yr of treatment.

Phosphorus removals at Flushing Meadows, Arizona, (134) were found to be basically dependent upon the distance traveled by the wastewater through the soil. The chief removal mechanism was precipitation as calcium phosphate or magnesium ammonium phosphate, since the soils tested contained little iron, aluminum oxides, or other phosphate fixing materials. Underground travel distances of 30 ft produced 50 percent reduction; distances of several hundred feet were found to be sufficient for 90 percent phosphorus removal. However, the capacity of the soil to remove phosphorus decreased after the start of the project, holding stable at approximately 50 percent removal.

The fact that soils gradually lose their capacity to adsorb phosphates over long-term application is substantiated by Barrow's detailed analysis of this phenomenon (68). He concluded that previously applied phosphate had been converted to a form that was occupying phosphate adsorption sites, thus reducing the capacity of the soil to further adsorb phosphate.

Continuing studies at Lake George, New York (47), have shown that significant amounts of nitrates appear to reach the waters tributary to the lake. However, the wastewater land treatment system appears to remove essentially all phosphorus, thus reducing the potential for algal bloom in the lake.

Dugan et al. (352), in their work on land disposal of wastewater in Hawaii, noted that phosphorus removals of over 95 percent within a 5-ft depth of percolation were obtained when secondary wastewater was applied to grassed areas.

Suspended solids are removed very effectively by land application systems. In fact, one problem encountered in the spreading of wastewater is that nearly all suspended solids are filtered out in the top few inches of soil. This can cause clogging of soil pores and reduction of infiltration rates. There is no danger of groundwater contamination in applied wastewater from suspended solids. Literature data on suspended solids are not extensive, because experience has shown land application to be capable of removing virturally all solids at the surface. A study at Whittier Narrows, California (1233), showed suspended solids removal, due to percolation, of 95 percent. At Flushing Meadows (134), the suspended solids concentration of the percolate was essentially zero, even though the solids in the wastewater applied reached 100 mg/ℓ. Similar findings were reported at Lake George (47), where virtually all BOD and suspended solids were removed from percolated effluent.

Results from a spray irrigation and runoff system used to dispose of a cannery waste (87) showed that even with a runoff-type system, suspended solids removals averaged 97 percent. Dugan et al. (352) reported similar high suspended solids removals with application of secondary effluent to Bermuda grass in Hawaii.

ELEMENTAL CONTAMINANTS

Municipal wastewater and sludge contain small amounts of nearly all metals. The degree to which a particular soil will protect underlying groundwater through removal of contaminants is primarily determined by the chemical and physical composition of the soil. Removal can occur through such processes as precipitation of solid phases, ion exchange, and adsorption. These processes are in turn controlled by soil pH, the oxidation/reduction potential, clay content, the presence and type of organic material, and the extent of soil saturation.

The general nature of reactions of sewage wastes with soil is well known. With time, wastes applied to land are broken down, and the dissolved constituents become part of the soil solution. Released cations can exchange with those already on exchange sites in the soil. Metals as ions or in the colloidal state can be adsorbed onto soil surfaces. When the levels of ions in solution exceed the solubility of corresponding solid phase compounds and minerals, those compounds can precipitate. When the solubility of solid phase compounds and minerals exceeds the levels of corresponding ions in solution, the compounds can dissolve. Constituents are also ingested by soil microorganisms and incorporated into soil organic matter.

Ions that are not removed by any of these processes but that remain in the soil solution are available for uptake by plant roots or leaching by water moving through the soil profile. Lindsay (798) studied the composition of the soil solution, concluding that it is controlled by the solubilities of solid phases. Thus, precipitation and dissolution reactions determine the activity of ions in solution, which in turn governs ion exchange.

Lindsay (798) also recognized the importance of the formation of metal-organic complexes and chelates in increasing the solubility and mobility of metals in soils. Brown (151) stressed this point. He found that it is misleading to predict the availability of metals in soils from their solubility in distilled water. He cites evidence that plant uptake of metal ions could not be predicted by the water solubility of the solid compounds. This may be because the soil solution in the vicinity of roots, unlike distilled water, is mildly acidic and contains organic metal-complexing agents (798).

All of the trace elements for which water quality criteria have been established may occur as either soluble or insoluble metal-organic complexes (1323). Low molecular weight organic molecules tend to increase the penetration of complexed metal ions into the soil, while high molecular weight molecules and their complexed ions may be filtered out by the soil (133). The chemistry of the metal organic complexes is complex, and present knowledge of organic forms of the elements is insufficient to generalize. Where concentrations of trace elements in soil solutions are in excess of those predicted from inorganic solubility product considerations, the element is thought to occur in organic form. Because of this lack of knowledge, the literature discusses mainly the chemistry of inorganic forms. This is unfortunate, because most of the metal ions in wastewater and sludge probably occur in complexed form.

Most metals are less mobile under aerobic and basic soil conditions than they are under anaerobic and acid conditions (133). The neutral water extract of soils contains less heavy metals than acidified extracts of the same soils, indicating the importance of the pH factor in influencing the mobility of metals (1173). Lindsay (798) cites evidence that for zinc and copper there is a 100-fold increase in ionic activity for each unit decrease in soil pH. Normally insoluble metals may become mobilized in the event of a change in the characteristics of percolating water, such that acidic or anaerobic conditions occur.

The removal of metals by ion exchange or adsorption depends upon the availability of exchange and adsorption sites in the soil as well as on the factors (ionic activity and pH) just discussed. Therefore, the clay content of the soil is important; clay soils provide more exchange and adsorption capacity than sands and gravels (1173). A correlation also exists between the form in which the elements occur in solution and their removal by exchange or adsorption. In general, elements that occur in solution as anions or neutral molecules pass through soils more readily than do elements that occur as cations. Inorganic arsenic, selenium, and fluorine in aerated soils occur as anions or neutral molecules. Although there are exceptions (depending upon the chemistry of the system), inorganic cadmium, copper, chromium, lead, mercury, silver, and zinc most commonly occur in fresh waters and soil solutions in the inorganic form as cations (1323).

Many conclusions in the literature lack substantiating experimental data but are based on a knowledge of chemistry. The use of sanitary sludge for land reclamation projects may increase the toxic metal content of the soil. The work of Lejcher and Kunkle (776) showed significant reductions in the leachate of iron, aluminum, copper, manganese, and sulfate combined with increases in cadmium, zinc, and chromium when 304 tons dry wt/ac were applied. In this experiment, the resulting pH of the surface soils was 6.2, still low enough to account for the mobilization of the metals. Higher application rates or the utilization of sludges with higher buffering capacities may help immobilize the metals that showed increases. The work by Sopper et al. (1299) revealed similar results.

Bernard (106) considered metal concentrations in sludge a prime deterrent to landfilling as a disposal method, but he concluded that metals tend to concentrate in the upper soil layer rather than leaching through the soil profile. Limited monitoring by Lofy (ongoing research) at eight landfill sites accepting wastewater sludge indicated that there was measurable migration of certain metals to considerable distances from the disposal area. Lead was particularly mobile. Preliminary indications are that many metals, such as chromium, are strongly attenuated by the majority of soils studied.

Prothero (1124) studied five sites where sludge is disposed of on land. The heaviest application rate at a site was 500 dry metric tons of sludge/ha. Water samples were collected from wells in close proximity to the sludge disposal locations and were analyzed for cadmium, chromium, manganese, nickel, and zinc. The trace element content of the applied sludge fell within ranges generally reported for municipal sludges, except for the chromium content of one sludge (17,700 ppm). Results showed the cadmium, chromium, and nickel contents of the well waters were generally undetectable while the copper, manganese, and zinc concentrations did not exceed 0.3, 0.01, and 0.04 ppm, respectively. There was an exceptionally high manganese content (1.2 ppm) at one site, however, which was apparently due to strong reducing conditions. The other well samples were below maximum allowable drinking water concentrations set by the Public Health Service for each metal.

Walker (1440) considered groundwater contamination by heavy metals at a sludge entrenchment site. For 19 months after sludge was buried at 350 to 500 tons/ac dry sludge solids, no metals were detected in groundwater observation wells.

Data from actual land application operations, which included arsenic and selenium determinations, were not found. Predictions of removal of these metals based on chemical properties are also hindered by limited understanding of the reactions involved.

Only one reference was located that provided experimental data on cadmium in relation to wastewater applied to land. At Flushing Meadows, Bouwer et al. (134) found that cadmium in wastewater applied to land in shallow basins showed very little change due to migration through the soil. The cadmium concentration dropped only slightly from 7.7 $\mu g/\ell$ to 7.2 $\mu g/\ell$. Aerobic conditions and alkaline pH prevailed in the soil studied. A study (1323) by the California State Water Resources Control Board (CSWRCB) reviewed available data on soil reduction of cadmium in effluents, concluding that information is not sufficiently detailed to allow adequate evaluation of cadmium concentrations in water reaching groundwater basins. According to the study, it must be demonstrated that a particular soil is able to reduce the concentration of cadmium to a level that is acceptable for drinking water. Otherwise, the study advised against the use of wastewater effluents with concentrations above this limit for groundwater recharge operations involving percolation through soil.

Bouwer's work at Flushing Meadows (134) found that the copper concentration of applied wastewater was reduced about 86 percent by passage through the soil. This removal occurred rapidly, usually in the first 30 ft of downward flow.

Iron and manganese in well-oxidized soils are characterized by the formation of highly insoluble oxides and hydroxides. However, at low pH and under reducing conditions, these metals can be solubilized and become mobile in the soil as Fe^{2+} and Mn^{2+}. Amramy (26) conducted a study of sewage lagoon effluent spreading on sand dunes. He found that after a subsurface travel distance of 8 m, the concentration of iron in the wastewater actually increased from 0.28 mg/ℓ to 0.57 mg/ℓ. The manganese concentration increased from 0.08 mg/ℓ to 0.19 mg/ℓ after 25 ft of travel through sand. Wesner and Baier (1484) found a similar phenomenon when tracing the underground movement of wastewater after injection. Over a travel distance of 400 or 500 ft, the concentration of iron did not change. Manganese concentrations, on the other hand, increased up to 300 percent in the first 100 ft of subsurface travel. Anaerobic conditions, favoring formation of soluble manganese compounds, are mentioned as the possible cause of the large increase. These conditions are most likely caused by biological oxidation depleting oxygen near the point of injection, thus causing the reduction of manganese to soluble forms. However, after greater travel distances and the return to aerobic conditions, the manganese may revert to insoluble compounds and be removed from the migrating water.

Ragone et al. (1129) reported on deep well recharge experiments conducted in Nassau County, Long Island. Tertiary effluent was recharged by a deep well into the Magothy aquifer, the primary water supply source for Nassau County. As of September 1972, 12 recharge tests had been run since the inception of the recharge program in September 1968. Although the iron concentrations of reclaimed and native water averaged 0.44 mg/ℓ and 0.24 mg/ℓ, respectively, the iron concentration of the mixed (native and reclaimed) water at times exceeded 3 mg/ℓ. The authors mentioned several sources that could account for the increase in iron concentration, but the most probable source was the pyrite native to the Magothy aquifer. During recharge, the natural reducing condition in the aquifer was replaced by a progressively more oxidizing environment. The initial response to this change was the oxidation of pyrite, which released Fe^{+2}, SO_4^{-2}, and H^+ into solution. Eventually, ferric hydroxide precipitated, and the Fe^{+2} concentration decreased. The exact oxidation mechanism apparently involved inorganic and/or organic constituents in the reclaimed water, because water from the public potable water supply system caused no increase in iron concentration when injected into the aquifer.

The Flushing Meadows study by Bouwer et al. (134) contained data on the reduction of lead by soil infiltration. Bouwer found that the wastewater concentration of lead decreased by 20 percent after significant travel distance underground. Apparently, a small portion of the lead was tied up rapidly (within 50 ft), while the majority was unaffected by further travel. It should be noted, however, that the soil examined in these experiments was a sand with limited adsorption and exchange capacity for many trace elements.

Both mercury and zinc form insoluble compounds in soil, lowering the activity of the ions in solution so that little movement occurs. Mercury is particularly insoluble as phosphate, carbonate, or sulfide. However, under low pH conditions, the metals may become mobilized. They can also form soluble complexes that affect their mobility under certain circumstances (798). At Flushing Meadows (134), underground percolation through 100 ft of sand produced 40 percent removal of mercury and approximately 58 percent removal of zinc. Further travel produced no further removal of mercury but reduced the zinc concentration to about 20 percent of that in the applied wastewater. The final concentrations of mercury and zinc were .00014 mg/ℓ and .037 mg/ℓ, respectively.

In conclusion, the information garnered in this study of the literature is inadequate to define fully the chemical behavior of elemental contaminants in the soil and their fate as they percolate through the unsaturated zone. The CSWRCB recommended, on the basis of its literature review (1323), that wastes containing concentrations of certain metals above those acceptable for drinking water supplies should not be applied to land, unless it can be demonstrated that contamination of groundwater does not occur.

BIOCIDAL CONTAMINANTS

Use of chemical pesticides in agriculture generated many studies of the potential harmful effects of these compounds on land, crops, surface water, and groundwater. However, there is little data available on biocides in municipal wastewater or on the potential dangers of groundwater contamination through land wastewater application operations. This lack of information is understandable considering the minimal role that municipal wastewater plays in transporting biocides to the land. Biocides come in contact with the land through a number of activities, primarily by direct application to the land for pest control. Return irrigation water, and spills and wastes from pesticide manufacturing operations also bring biocides in contact with the land.

The reactions of pesticides with soil has received limited attention in the literature. Volatilization, chemical degradation, and absorption by plant roots and seeds apparently remove a small portion of pesticides reaching the soil. A more significant process for pesticide removal may be microbiological degradation. Although this process is very slow in some cases, often taking several years, microbiological degradation accounts for the breakdown of a remarkable variety of organic compounds. Pesticides that are not removed from the soil column or broken down by these processes may be available for leaching into groundwaters.

Gerakis and Sficas (464) reviewed the literature on pesticide degradation and leaching. They reported that the most important factors involved in these two processes are soil temperature and moisture, organic matter and clay content, soil management practices, pH, and species and population density of microorganisms present. The presence of organic matter and clay in the soil appears to be positively correlated with adsorption of pesticides onto soil particles.

Van Bladel and Moreale (1417) studied herbicide adsorption onto clay minerals. They found that adsorption increased with the polarizing power of the exchangeable cation, and concluded that adsorption appears to be one of the most important factors in reducing pesticide removal from soil layers by leaching.

Laboratory studies of land-applied sludge were conducted by the University of Nebraska (1474) to determine degradation rates in groundwater for selected herbicides. The study indicates that herbicide degradation was much slower in groundwater than in soil. O'Connor and Anderson (1014) analyzed factors affecting adsorption of the herbicide 2,4, 5-T on four soils in the western United States. The study found that organic matter contributed to adsorption, while oxides of iron and aluminum did not.

Gerakis and Sficas (464) cite evidence that pesticides differ in their mobilities in the same soil. One study showed that (1) acidic compounds are relatively mobile, (2) phenyl ureas and triazines are of intermediate to low mobility, and (3) organochlorine compounds and organic cations are least mobile. Further data that were reviewed supported the conclusion that under normal agricultural practices and rainfall, it is very unlikely that pesticides may be leached deeply enough and in such quantities as to cause appreciable contamination of groundwaters.

A California State Water Resources Control Board report (1323) mentioned that pesticides are adsorbed by soil clays, iron aluminum oxides, and especially by organic colloids, and that they are susceptible to microbial decomposition. However, the amount of biocides in average municipal wastewater and sludges was found to be so minimal that the spreading of municipal waste on land offers extremely low potential for groundwater biocide contamination.

SYNTHETIC/ORGANIC CONTAMINANTS

One of the most intensely debated questions regarding land application for treatment and/or disposal of municipal wastes concerns the problem of residual organic contaminants. Refractory organic compounds may survive conventional treatment processes and penetrate through the soil to contaminate groundwater supplies. The controversy centers around the fate of residual organics within the soil systems, including such issues as the synergistic effects between organics and inorganics or other groundwater and soil constituents, or conversion of safe organics to hazardous compounds in the soil. Despite this controversy, no literature was found concerning groundwater pollution by the synthetic/organic contaminants in municipal wastes as a result of land application.

The absence of literature concerning the movement through soil of synthetic/organic contaminants in applied sludge is not surprising, since the specific organic makeup of sludge is unknown. Some of the chemicals of concern (PCB's, polycyclic aromatics, and other chlorinated hydrocarbons, etc.) have low solubilities in water in comparison with vapor pressures. As a result, there is a distinct possibility of vaporization when sludges containing these chemicals are applied to the land surface.

The California State Water Resources Control Board (1323) cited a study carried out in Colorado that compared the nature of the soluble organic material in the soil profiles under a feedlot and under grassland with selected ground, well, and river waters. It was concluded that the major portion of the soluble materials in all the waters was polymeric. The soluble organics under grassland were essentially the same as those under the feedlot, although phenols were present in greater abundance in the manure and surface soil of the feedlots. About 13 percent of the soluble material in the soil profiles was carbohydrate (polysaccharides), and much of the remainder, based on IR spectra and reductive degradation procedures, appeared to be polymerized aromatic structures. This report by the CSWRCB interpreted these observations to indicate that the soluble organics under wastewater-treated soils would be similar to those under feedlot manure or grassland.

BIOLOGICAL CONTAMINANTS

Most available data suggest that virus, bacteria, and other biological pathogens present in wastewater and sludge are removed or inactivated by percolation through soil.

The California State Water Resources Control Board study (1323) provides a summary of the fate of viruses, bacteria, protozoa, and parasitic worms in wastewaters applied to land. The summary states that most of these pathogens prefer warm-blooded animals as their habitat and do not flourish in the soil environment. When introduced into soils, the pathogens do not compete well with the vast number and variety of normal soil inhabitants and are subject to attack by antagonistic soil species. The time necessary for their ultimate destruction varies, according to species and environmental conditions. A compilation of pathogen survival data in the literature is shown in Table 92 below.

TABLE 92 . SURVIVAL OF PATHOGENS IN SOILS (1323)

Ascaris lumbricoides ova	2.5 - 7 years
Endamoeba histolytica cysts	8 days
Salmonella species	6 hours
Coliform group organisms	133 - 147 days
Q-fever organisms	148 days
Brucella abortus	30 - 100 days
Tuberculosis bacteria	6 months
Enteroviruses	12 days

The most persistent pathogens in soils appear to be ova, cysts, and spore-forming bacteria. The survival of enteric viruses in soil has not been thoroughly studied. The dependence of enteric viruses on specific host organisms for reproduction suggests that they would not multiply and would not be expected to survive for a long period of time, although survival may, at times, be long enough to cause public health concern.

Two and a half years of continuous observation was conducted of wastewater reclamation by landspreading in Lodi, California (1175). It was found that the MPN of coliform group organisms, which averaged $1.9 \times 10^8/100\ m\ell$ in the wastewater, was consistently reduced to less than $1/100\ m\ell$ after 4 to 7 ft of soil travel. The average percolation rate was 0.3 ft/day in coarse-textured Hanford sandy loam. It was observed that the number of coliforms penetrating 1 ft or more was essentially independent of the coliform concentration of the wastewater.

At Whittier Narrows (892), percolation tests showed that vertical percolation of wastewater through 4 to 7 ft of soil is an effective method of removing bacteria of fecal origin, despite heavy growth of coliforms of soil origin. The formation of an organic-microbial slime layer at the water-soil interface was found to increase the efficiency of the filtering action.

Lejcher and Kunkle (776) reported, with regard to biological pathogens, that no fecal coliform bacteria were found in the runoff from the sludge-treated plots (treated at a rate of 78 to 304 dry tons of sludge/ha) during the year immediately following application. They concluded that it appeared unlikely that pathogens would survive in the runoff. Limited transfer of microbial cells through the soil profile and rapid die-off of the pathogens in the soil matrix suggested low-risk conditions (619, 917).

Research by Bernard (106) on the disposal of sewage sludge to sanitary landfills concluded that pathogens tend to concentrate in the upper soil layer, rather than leaching through the soil profile. In contrast, Lofy (ongoing research) found that fecal coliform and fecal streptococcus are particularly mobile and can travel considerable distances.

Results from studies at Flushing Meadows, Arizona (134), show that fecal coliform density was reduced significantly in the first 2 or 3 ft of travel. Bouwer found that fecal coliform density at a particular depth tended to decrease with increased flooding time. The peak bacteria density invariably appeared immediately after flooding was resumed. The concentration of fecal coliforms was consistently decreased to less than 10/100 mℓ after 100 ft, and to 0/100 mℓ after 300 ft of travel.

A project at Santee, California (1215), is famous for its pioneering work in the reclamation of domestic sewage for recreational lakes. Travel of secondary effluent through 1,500 ft of very coarse sand was sufficient to remove all fecal coliforms. Sampling showed that most of the coliforms were removed in the first 200 ft.

At Orange County, California (1484), tests conducted on a well injection system showed coliform organisms 30 m from the injection well, but none approximately 80 m from the well. The results indicated that fecal coliforms are more easily removed by underground travel than other coliforms. Some of the other coliforms may have been supported by nutrients in the effluent.

Results of percolation tests at Lake George, New York (1129), showed again that percolation of applied secondary effluent through 5 to 10 ft of soil in two different beds was sufficient to remove essentially all coliform organisms.

Browning and Mankin (155) reported an unusual case of disease outbreak due to contamination of groundwater well supplies by land application of treated sewage. In Madera, California, undisinfected secondary effluent was used to irrigate a pasture located adjacent to a deep well drawing part of the city water supply. The wastewater migrated through gopher holes, filling a construction pit around the well, and eventually flowed into the well itself.

On the basis of experience and results of full-scale, long-term wastewater reclamation studies, the CSWRCB (1323) concluded that, although soil is an excellent media for removing bacteria, a small fraction of the fecal coliform bacteria therein may reach groundwater reservoirs at high percolation rates.

Horizontal travel of viable fecal coliform bacteria in the aquifer does not appear to occur to a significant degree. The available data on horizontal travel, however, are inconclusive. Further investigation of the transport and survival of pathogenic bacteria in groundwater, therefore, is required.

Research shows that travel through soil removes significant amounts of viruses, primarily through adsorption. Adsorption is influenced by the pH and ionic strength of the soil solution. The available information indicates that adsorption of virus by soil is nearly complete at pH 7 or less, but decreases as the pH value increases above 7. This is mainly because the overall electric charge surrounding both the virus and soil particles becomes increasingly negative as pH levels increase and, therefore, mutual repulsion occurs (1323).

It also appears that increasing cationic strength of the percolating water or soil solution increases virus removal. At the pH values normally encountered in wastewater, viruses are slightly negatively charged. The presence of calcium, magnesium, sodium, aluminum, and other positive ions in the soil solution decreases the potential for negatively charged soil and virus particles to repel each other. This results in the formation of soil-cation-virus bridges that immobilize virions (1323).

The ionic strength in percolating wastewater is usually sufficient so that it does not limit adsorption. In circumstances where ionic strength is significantly decreased, however, desorption of adsorbed viruses may occur. Organic matter in wastewater can also compete with viruses for adsorption sites. In laboratory studies, when virus-adsorbed clay particles were washed with distilled water, an essentially complete desorption and reactivation of viruses took place. In field conditions, other mechanisms in soil systems may inactivate or destroy adsorbed viruses before they are subject to desorption (1323). In addition to pH and ionic strength, the clay and organic matter content of soil evidently influences adsorption to some degree. In general, soils of higher clay and/or organic matter content are more effective in adsorbing viruses (1323).

Definitive work on virus interaction with soil was conducted at Santee, California, where extensive studies showed that percolation through several hundred feet of soil consistently removed all virus from secondary effluent (1215).

Other studies also supported the conclusion that soil effectively removes viruses. Viral analyses in Hawaii by Dugan et al. (352) showed that test soils in 5-ft lysimeters were completely effective in removing viruses. Brief tests at Whittier Narrows, California (892), achieved complete removals of Sabin Type III polio-virus vaccine. Although 250 plaque-forming units (PFU) of enteric viruses/ℓ were present in the applied wastewater, no measurable concentrations were found below 2 ft in the percolate.

In 1974 at Flushing Meadows (134), virus analyses were performed bimonthly to determine the fate of viruses in the soil system. Secondary effluent was allowed to infiltrate into six parallel horizontal basins consisting of 60 to 90 cm of fine loamy sand underlain by several coarse sand and gravel layers to a depth of 75 m, where a clay layer begins. Observation wells were installed in line across the basin area. No viruses were detected in any of the wells at any time during each flooding period. Gilbert et al. (475) stated that the failure to detect viruses in the wells indicates that the virus count was reduced by at least 99.99 percent within 3 to 9 m of basin soil.

Romero (1175) reviewed the studies performed for the Department of the Army on sands ranging in classification from silty sand to coarse-granite alluvium. Results indicated that the bacterophages T1, T2, and 65 are more effectively retained in the finer sands, particularly in those containing a relatively high percentage of clay and silt. Virus removal was shown to increase with decreasing particle size. The greatest percentage of removal took place in the uppermost portion of the sand columns tested. It was shown that for a well-sorted sand of particle size averaging 0.12 mm, the removal efficiency in 2 ft of penetration was 99.999 percent.

Young and Burbank (1548) described studies of virus removal in Hawaiian soils. In the laboratory, columns of various types of Hawaiian soil were subjected intermittently to percolating water with a known concentration of virus (coliphage T4B II mutant, and polio virus Type II (Lansing) H8), simulating the action of cesspool leaching. The effluent from each soil column was analyzed for viral content. Coliphage T4 was applied to slightly acid soils (pH 5-6) at a concentration of 2.5×10^6/mℓ. Percolation through 2.5 to 6 in of soil was 100 percent effective in retention of the virus. Slightly alkaline soil was less effective, removing only 67 percent of applied coliphage and 35 percent of applied polio virus in 15 in. Removal of polio virus Type II was less complete; 6-in columns were able to effect only 99.3 percent removal with an initial feed concentration of 1.5×10^5 pfu/mℓ.

Wellings et al. (1479) found that virus can be isolated at the 6.5-m level below a spray irrigation field. Another study by Wellings et al. (1477) measured virus migration through the ground from chlorinated packaged plant effluent applied to a cypress dome. Both horizontal and vertical migration was detected at distances of approximately 7 m for polio and coxsackie viruses. Wells beyond that distance showed no virus. The survival of virus within the dome was at least 28 days.

At Fort Devins, Massachusetts, where a land application site has been in operation for over 30 years, Schaub et al. (1210) studied the removal of bacteria from unchlorinated primary effluent applied to soil cells. Using tracer f2 bacteriophage and the enteroviruses polio virus I and EMC virus, it was demonstrated that tracer bacteriophage penetrated into the groundwater along with the percolating wastewater. The concentration in the groundwater stabilized at almost 50 percent of the applied virus concentration. The tracer and enteroviruses were sporadically detected at horizontal distances up to 600 ft from the application point.

Lance et al. (750) passed secondary sewage effluent containing 3 x 10^4 pfu/mℓ polio-virus Type I (LSc) through 250 cm-long columns packed with calcareous sand from an area in the Salt River bed used for groundwater recharge of secondary sewage effluent. Viruses were not detected in 1-mℓ samples extracted from columns below the 160-cm level, but were detected in 5 of 43 100-mℓ samples of the column drainage water. Most of the viruses were adsorbed in the top 5 cm of soil. Virus removal was not affected by the infiltration rate, which varied between 15 and 55 cm/day. Flooding a column continuously for 27 days did not saturate the top few centimeters of soil with viruses and did not seem to affect virus movement. Flooding with deionized water caused virus desorption from the soil and increased virus movement through the columns. Drying the soil for one day between applying the virus and flooding with deionized water greatly reduced desorption, and drying for five days totally prevented desorption. The investigators concluded that large reductions (99.99 percent or more) of virus are expected after passage of secondary effluent through 250 cm or more of calcareous sand, unless heavy rains fall within one day after application of sewage.

PATHWAYS TO MAN

AIR

INTRODUCTION

Health-impairing contaminants contained in wastewater are most likely to return to man by way of surface or groundwater migration; however, some contaminants may also be transmitted through the atmosphere. Aerosolized particles ranging in size from 1 to 40 microns are generated as an incidental by-product of various wastewater treatment processes and may contain health-impairing contaminants that can be directly inhaled by man. Glaser and Ledbetter (481) found that 40 percent of the particles aerosolized during activated sludge treatment were smaller than 10 microns. Larger particles pose a minimal hazard to respiratory systems but may be ingested; however, aerosols of 10 microns or less will be captured in the upper respiratory tract, transported through the pharynx, and will enter the digestive tract.

Activated sludge, trickling filters, and spray irrigation of wastewaters and liquid sludges are considered to be the major potential sources of aerosols. In addition, vaporized contaminants are emitted from sludge incinerators and furnaces used for activated carbon regeneration.

Bubbles are produced in wastewater by natural biological action or by mechanical aeration during activated sludge and similar processes. These bubbles burst at the surface, ejecting contaminant-bearing water droplets into the air. When wastewater is sprayed, as in land application, large numbers of particles are released into the atmosphere. Aerosols containing wastewater contaminants are also produced above bodies of water into which wastewater effluent is released as a result of wind spray, breaking waves, splashing rain, or bursting bubbles.

Certain contaminants are floatable and tend to concentrate in a layer on the water surface. Particles that are aerosolized from this layer may contain high concentrations of these contaminants relative to their concentrations in the bulk of the liquid. For example, Blanchard and Syzdik (125) found the bacterial concentration of *Serratia marcescens* in drops from bursting bubbles to be from 10 to 1000 times higher than that of the tap water in which the bubbles formed. These investigators suggested that DDT and several species or organisms could be expected to concentrate in surface films and be transferred to the atmosphere.

Direct vaporization of certain contaminants also occurs. For instance, low solubility contaminants such as pesticides, polychlorinated biphenyls, or hydrocarbons have high evaporation rates from water bodies to the atmosphere, according to a paper cited by Mackay and Leinonen (827).

The viability of biological aerosols from sewage treatment plants is affected by distance traveled, deposition, dispersion, and hostile environmental conditions.

One study revealed the correlation between human respiratory difficulty and high concentrations of plankton in the sea. This study has often been cited (125) as indicative of the airborne, onshore movement of marine organisms. The "blowback" of microorganisms rising to the surface from submarine sewage outfalls may also have a significant, localized impact on public health.

A considerable body of information has been developed in medical literature pertaining to the movement of pathogenic bacteria through air. In scientific literature topics studied include atmospheric physics and dispersion modeling, aerosol physics, liquid surface physics, dissolved air and froth flotation technology, and other subjects indirectly related to potential atmospheric pathways of contaminants from wastewater treatment systems to man. Very little of this work, however, deals directly with the subject of atmospheric pathways of contaminants from wastewater to man. Available literature is listed in Table 93. Research in this area has been principally concerned with the transport of biological pathogens.

ELEMENTAL CONTAMINANTS

For most elemental impurities, direct vaporization will not occur. Few references were found in the literature reviewed that dealt directly with vaporization or aerosolization from wastewater systems, although Haque and Freed (538) included a discussion of the thermodynamics involved.

The atmospheric transport of mercury and dimethyl-mercury has been studied and reviewed by Jernelov et al. (645), with theoretical calculations performed by Baughman et al. (78). On the basis of Baughman's calculations, it can be assumed that the evaporative loss of these substances from aqueous solutions may pose a problem under turbulent conditions. Elemental mercury appears to be volatilized twice as rapidly as the dimethyl form.

Soldano et al. (1295) measured organic and elemental mercury concentrations in cities with central sewage facilities. A broad range of mercury concentrations was detected in the vicinity of treatment facilities. Numerous measurements were in the range of 10^3 ng/m^3, but values as low as 0.125 ng/m^3

TABLE 93. LITERATURE REVIEWED PERTAINING TO ATMOSPHERIC TRANSPORT

Contaminant	Reference Number
Water Quality Parameters	481
Elemental Contaminants	
Boron	1113
Mercury	78, 538, 645, 1295, 1302
Selenium	1114
Other (general)	538
Biocidal Contaminants	
Chlorinated hydrocarbons	1532
DDT	553, 1472, 1551
Dieldrin	553, 1551
2,4D	553
Synthetic/Organic Contaminants	1219, 1564
Biological Contaminants	
Bacteria	79, 125, 238, 400, 481, 545, 568, 624, 649, 685, 765, 766, 792, 936, 983, 1069, 1073, 1099, 1100, 1234, 1302, 1304, 1463, 1560
Coliforms	2, 400, 568, 685, 983, 1099, 1100, 1234, 1304
Hepatitis virus	1151
Parasitic worms	1234
Protozoa	1234
Salmonella	568, 1234
Shigella	568, 1234

TABLE 93 (continued)

Contaminant	Reference Number
Virus	79, 238, 400, 649, 792, 1261, 1302, 1304
Other (general)	125, 238, 481, 568, 765, 766, 983, 1073, 1234

were also found. The atmospheric concentration of elemental mercury fell off sharply with increasing distance from the sewage plant, while the concentration of organic mercury rose with increasing distance. The concentration maxima of organic mercury at any particular plant was related to population size of the area served. The authors concluded that sewage treatment systems concentrate and re-emit elemental mercury after some of it has been converted into the more volatile organic forms by bacterial action.

BIOCIDAL CONTAMINANTS

Biocides tend to concentrate in the liquid surface and aerosol fractions. However, the threat of adverse health effects from aerosolized biocides contributed by sanitary waste systems appears small in relation to that posed by other sources of these contaminants. It is therefore not surprising that only indirectly related research is available on the transport of biocides from sanitary waste systems through the atmosphere. Information on pesticides in rainfall and runoff is provided by Weibel et al. (1472), and factors affecting the volatility of DDT, dieldrin, and 2, 4-D from leaf and glass surfaces have been examined by Hee et al. (553). Young and Heesen (1532) conducted a study of the atmospheric transport of chlorinated hydrocarbons to the waters of three southern California harbors. They determined that atmospheric transport accounted for less of the total concentrations of p,p'-DDT and PCB 1254 than surface runoff.

SYNTHETIC/ORGANIC CONTAMINANTS

In general, insoluble or limited solubility organic materials are concentrated at the air/water interface, in turn producing a concentrated aerosol. The primary significance of this surface concentration of organic materials and aerosols lies in the effect that these materials may have on the survival of biological aerosols. Webb (1463) has conducted experiments showing enhanced survival of several species of bacteria when amino acids, long chain protein degradates, some sugars, and polyhydroxycyclohexanes were added to bacterial suspensions before aerosolization. Hatch and Wolochow (545) have also reviewed several aspects of this problem, concluding that most of the organic compounds shown to be the best protective agents for pathogen survival are sugars or polyhydricalcohols.

BIOLOGICAL CONTAMINANTS

A number of recent studies have examined the airborne bacterial levels adjacent to treatment plant and land irrigation sites. However, no correlation has yet been made between specific atmospheric organism levels and the incidence of disease (238). In their 1975 review of the literature, Hickey and Reist (568) found only one survey that had a direct bearing on increased disease rates associated with occupational exposure to viable aerosols in wastewater treatment operations. In this survey, the rate of pneumonia incidence was identical for both water purification plant and sewage plant workers, but the incidence of flu and colds was higher among sewage workers by factors of 50 to 28 percent, respectively.

The literature on airborne levels of micro-organisms is difficult to interpret for several reasons. For instance, sampling methods are not standardized, and a suitable indicator organism is lacking. Also, there has been no research to date suggesting that the traditional water quality indicator organisms, the coliforms, provide a reliable indication of pathogens present in aerosolized particles. In fact, the half-life of airborne coliforms was found to be shorter than that of the total bacteria in studies of the downwind dispersion of bacterial aerosols from activated sludge tanks performed by Ledbetter and Randall (766). Pavoni and Tittlebaum (1069) cited a study that found the survival rate of enterobacteriaceae to be only 13 percent of that recorded for the total bacterial aerosol emanating from an activated sludge unit. A second study cited by Pavoni and Tittlebaum (1069) indicated that capsulated organisms (klebsiella, aerobacter) are better able to survive in the air than acapsulate organisms (escherichia). The suitability of coliforms and coliphages as animal virus indicators was the subject of research by Fannin et al. (400). Evidence collected from both trickling filter and activated sludge plants showed coliforms to be far less stable than coliphages in the airborne state. On the basis of this evidence, it was suggested that coliphages may prove to be far more acceptable indicators of airborne animal viral contamination than coliforms.

Most research to date, however, has relied upon coliforms as primary indicator organisms to determine total bacterial concentrations. The bacterial concentrations at various distances from treatment units, which are reported in the literature by a number of investigators, are given in Table 94. This table represents a sample of the available literature on activated sludge and trickling filter plants.

The Adams and Spendlove paper (2) is unique in that it reports recovery of airborne coliforms as far as 0.8 mi downwind from a large trickling filter plant. In contrast, Hickey and Reist (568) cited one study in which no Escherichia coli was found beyond 100 ft, and a second study that reported no observable effects at greater than 150 ft downwind from a trickling filter. Napolitano and Rowe (983) offer data showing duplicate samples with coliform concentrations that differ by as much as two orders of magnitude.

Relatively few investigations involved the recovery of actual pathogens. According to research reviewed by Hickey and Reist (568), of the total bacteria recovered near aeration tanks, 10.5 percent were klebsiella, aerobacter or proteus organisms. Despite repeated long-term sampling, neither the shigella nor salmonella bacteria were recovered near the tanks, a finding that was expected considering the small numbers of these genera found in wastewater. Airborne Staphylococcus aureus, hemolytic streptococcus, mycobaterium, and acid-fast bacilli were also isolated.

A study conducted by Fannin et al. (400) evaluated animal virus concentrations. No animal viruses were found in aerosol emissions from activated sludge and trickling filter plants with sewage containing about 100 plaque-forming units (pfu)/ℓ. Airborne coliphage viruses and coliform bacteria were recovered at average levels of .23 to .3/m^3 and 210 colonies/m^3, respectively. As shown in Table 94, the average coliphage concentration decreased with distance from the source, but only the coliforms showed a statistically significant decrease. Fannin et al. compared the ratio of airborne coliphages to coliphages found in sewage with the level of animal viruses in sewage and calculated that the expected concentration of airborne animal viruses is 6.2-6.5 x 10^{-5}/m^3 (Most Probable Number).

In summary, the literature indicates that activated sludge and trickling filter units emit potentially hazardous bacterial aerosols and low levels of airborne animal viruses. The concentrations of viable organisms generally decrease rapidly with distance from the source, the rate of decrease depending on such variables as wind speed, relative humidity, solar radiation, air temperature, and obstructing vegetation.

TABLE 94. BACTERIA AND VIRUS CONCENTRATIONS IN AEROSOLS FROM TRICKLING FILTERS AND ACTIVATED SLUDGE UNITS

Researcher	Ref. No.	Aerosol Source	Sampling Distance Downwind	Organisms Sampled for	Levels Identified	Comments
Hickey and Reist (LR)*	568	Trickling filter	100 ft	*E. coli*	Unspecified	Max. dist. where recovered
			>3 ft	*E. coli*	6/cu ft	Max. level recovered at >3 ft
				A. aerogenes	5/cu ft	Max. level recovered at >3 ft
Ledbetter and Randall	966	Activated sludge	20 ft	Coliforms	5 colonies/ plate	EMB agar plates Subtracted upwind controls
			100 ft	Coliforms	1 colony/ plate	
Napolitano and Rowe	983	Activated sludge	0 ft	Coliforms	2.8, 21.8/ ft^3	50% particles <5 μ in diameter
			150 ft	Coliforms	0.2, 30.8/ cu ft^3	
		Trickling filter	0 ft	Coliforms	3.0, 29.8/ft^3	
			150 ft	Coliforms	0.2, 4.0/ft^3	
Adams & Spendlove	2	Trickling filter	4,224 ft (0.8 mi)	Coliforms	<1 ft^3 (3-4/m^3)	
			150 ft	Coliforms	3-26 ft^3	
			1,800 ft	Coliforms	0.2-3 ft^3	
Clark, et al. (LR)*	238	Trickling filter	150 ft	Coliforms	1.4-8.5/ft^3	

TABLE 94 (continued)

Researcher	Ref. No.	Aerosol Source	Sampling Distance Downwind	Organisms Sampled for	Levels Identified	Comments
Hickey and Reist (LR)*	568	Activated sludge	Immediately downwind	Total bacteria	$870/ft^3$	Andersen impactor
				Total bacteria	$1,170/ft^3$	All-glass impinger
				Klebsiella Aerobacter Proteus	10.5% of total bacteria	Known respiratory tract pathogens
Hickey and Reist (LR)*	568	Activated sludge	150 ft	Total bacteria	$3/ft^3$	Same as concentration 150 ft upwind. No observable effect at >150 ft downwind
Hickey and Reist (LR)*	568	Activated sludge	Unspecified	Staphylococcus aureus	Recovered five times as often downwind as upwind	
Hickey and Reist (LR)*	568	Enclosed activated sludge, force-ventilated through stacks	1,181 ft	Hemolytic streptococcus	Unspecified	Recovered inside and outside building
			98 ft	Mycobacterium	Unspecified	Recovered in exhaust stack
			72 ft	Acid-fast bacilli	Unspecified	Produced lesions in guinea pigs
Fannin, et al.	400	Trickling filter	$\leq$16.4 ft $\geq$16.4 ft	Coliphage virus	$.32/m^3$ $.24/m^3$	Average Most Probable Number values. Only coliforms show a statistically significant decrease with distance.
			$\leq$16.4 ft $\geq$16.4 ft	Coliphage virus	$.25/m^3$ $.16/m^3$	

TABLE 94 (continued)

Researcher	Ref. No.	Aerosol Source	Sampling Distance Downwind	Organisms Sampled for	Levels Identified	Comments
Fannin, et al.	400	Trickling filter (cont'd)	$\leq$16.4 ft $\geq$16.4 ft	Coliphage bacteria	210/m^3 14/m^3	
Fannin, et al.	400	Trickling filter and activated sludge	Variety of distances	Animal viruses	None recovered	Present in sewage at 100 pfu/ℓ

* LR - Literature Review

Poon (1099, 1100) used *Escherichia coli* to investigate the effects of different factors on the viability of airborne bacteria and to determine the mechanism involved in bacterial destruction. His studies indicate that the destruction increased in direct proportion to dryness of air and increased exponentially as temperature increased. Sodium chloride in bacterial aerosols affected the destruction by retarding water evaporation from within cells. Webb (1463) concurred that water evaporation is responsible for the destruction of bacterial aerosols in his study of *Serratia marcescens*, *Escherichia coli*, *Staphylococcus albus*, and *Bacillis subtilis*.

Both Poon and Webb conducted their research under laboratory conditions. Results of field studies have not always proved as consistent. The expected correlation between the number of colonies recovered downwind from activated sludge tanks and air temperature was not substantiated by Ledbetter and Randall (766). Higher rates of bacterial recovery were found to occur when wind speed or relative humidity was high. Data from these tests also revealed that the rate of bacterial die-off was characterized by two distinct phases: an extremely short and rapid initial die-off followed by stabilization, and a very slow die-off rate for the more resistant organisms.

A study cited by Hickey and Reist (568) indicates that the downwind travel distance of the viable aerosol from wastewater sprays increases as both relative humidity and wind speed increase, but will decrease with increasing ultraviolet radiation. It was estimated that coliform organisms may remain viable as far as 400 m downwind from the source under conditions of darkness, 100 percent relative humidity, and wind speed of 7 m/sec.

In recent years, spray irrigation has become popular for municipal wastewater and liquid sludges. Consequently, some studies have been made of the problem of biological aerosols generated by spray irrigation. Coliform bacteria were found in the air 350 m downwind from wastewater spray sprinklers by Katzenelson and Teltch (685). Their research program consisted of air sampling performed at various distances downwind from the line of irrigation. Bacterial colonies suspected to be of salmonella species were examined. Only one salmonella bacterium was found during the sampling, at a distance of 60 m. Salmonellae are relatively rare in wastewater, so this result was not surprising. However, the detection of even one pathogenic salmonella in the air was felt by the authors to have important public health implications. The authors also calculated, on the basis of their findings and normal breathing volume, that an individual working at a distance of 100 m from a wastewater sprinkler would inhale about 36 coliform bacteria every 10 min.

Sorber et al. (1302) calculated the health hazards posed by viral and bacterial aerosols released in spray irrigation of wastewater. Calculations were derived from a model developed to ascertain the effects of varying degrees of treatment and meteorological conditions on virus viability. On the basis of this model, the authors determined that within 10 min, up to 20 infectious units of airborne virus could be inhaled by an individual working 200 m downwind from the irrigation site. The 200-m distance provided no definitive safeguard, as significant numbers of enteric viruses were were estimated to survive beyond this point. Increased temperature, lower relative humidity, and sunlight reduced the viability of such pathogenic microorganisms; however, neither meteorological conditions or aerosol dilution by diffusion were judged to provide reliable pathogen reductions. The authors suggest instead that removals be effected either by treating wastewater before disposal or by allowing an adequate buffer zone around the spray irrigation site. An 800-m buffer zone would provide a reduction of two orders of magnitude in airborne virus concentration. Removal or destruction of wastewater-contained microorganisms by filtration or disinfection prior to spray application - which would achieve reductions of up to three orders of magnitude - was judged to be a more certain method of reducing the aerosol hazard.

A recent project reported by Shuval et al. (1261) demonstrated that spray irrigation of unchlorinated wastewater may in fact have severe implications for public health. The incidence of enteric communicable diseases at 77 agricultural settlements practicing spray irrigation of wastewater was significantly higher than at 130 kibbutzim studied, where no form of wastewater irrigation was practiced. In the communities using wastewater for crop irrigation, the incidence of shigellosis, salmonellosis, typhoid fever, and infectious hepatitis was two to four times higher than in the control communities. Significantly, no difference in the outbreak of disease caused by enteric virus was noted between test and control communities during the winter months, when no irrigation occurred. It was concluded that effective viral and bacterial disinfection is required if spray irrigation or land disposal of wastewater takes place near residential areas.

Research evidence indicates that biological pathogens are present in wastewater aerosols and that these pathogens may be inhaled by man with serious consequences to human health. However, precise information on the minimum dose of these organisms required to produce human infection is not yet available. This problem is one that ought to receive increasing attention as wastewater reuse for irrigation becomes more widespread.

CHEMICAL COAGULATION/FLOCCULATION FOLLOWED BY SOLIDS SEPARATION

Introduction

Chemical coagulation and flocculation, followed by clarification or filtration, is common water treatment practice for the treatment of surface waters. The primary purpose is to remove suspended and colloidal solids.

The overall process takes place in three distinct phases. Coagulation involves destabilization of the colloids by rapid mixing of the chemical coagulant with the water in some type of agitated rapid mix tank. Retention time in rapid mixing is very brief, on the order of a few minutes. Flocculation follows in which the wastewater is gently stirred with paddles, allowing the particles to collide and aggregate into larger flocs. Depending on temperature, concentration of the solids, and the type and dosage of coagulant, flocculation requires from 15 minutes to one hour. Clarification and/or filtration usually follows, to provide solids separation.

Commercially available flocculator-clarifier units (often called solids contact units) combine all three operations in a single compartmented tank. In a typical design, coagulant is fed and mixed with the wastewater at the influent pipe; flocculation occurs in a central cone-shaped skirt where a high floc concentration is maintained. Flow passing under the skirt passes up through a solids blanket and out over surface weirs. These units are particularly advantageous for lime softening of hard water, since the precipitated solids help seed the floc, growing larger crystals of precipitate to provide a thicker waste sludge. Recently, flocculator-clarifiers have been receiving wider application in the chemical treatment of industrial wastes and surface water supplies. The major advantages promoting their use are reduced space requirements and less costly installation. However, the unitized nature of construction generally results in a sacrifice of operating flexibility.

The primary substances used as coagulants are described below:

1. Aluminum sulfate + calcium carbonate
   $Al_2(SO_4)_3 + 3\ Ca(HCO_3)_2 = 3\ CaSO_4 + 2\ Al(OH)_3 + 6\ CO_2$

2. Aluminum sulfate + sodium aluminate:
$6NaAlO_2 + Al_4(SO_4)_3 \cdot 18\ H_2O + 8Al(OH)_3 + 3Na_2SO_4 + 6H_2O$

3. Aluminum chloride (used under exceptional circumstances only):
$2\ AlCl_3 + 3\ Ca(HCO_3)_2 = 2\ Al(OH)_3 + 3\ CACl_2 + 6\ CO_2$

4. Aluminum sulfate + hydrated lime:
$Al_2(SO_4)_3 + 3\ Ca(OH)_2 = 3\ CaSO_4 + 2\ Al(OH)_3$

5. Ferric sulfate:
$Fe_2(SO_4)_3 + 3\ Ca(HCO_3)_2 = 2\ Fe(OH)_3 + 3\ CaSO_4 + 6\ CO_2$

6. Ferric sulfate + hydrated lime:
$Fe_2(SO)_3 + 3\ Ca(OH)_2 = 2\ Fe(OH)_3 + 3\ CaSO_4$

7. Ferrous sulfate:
$FeSO_4 + Ca(HCO_3)_2 = Fe(OH)_2 + CaSO_4 + 2\ CO_2$

8. Ferrous sulfate + hydrated lime:
$FeSO_4 + Ca(OH)_2 = Fe(OH)_2 + CaSO_4$

9. Ferrous sulfate + chlorine
$2\ FeSO_4 + 3\ Ca(HCO_3)_2 + Cl_2 = 2\ Fe(OH)_3 + 2\ CaSO_4 + CaCl_2 + 6\ CO_2$

The most commonly used coagulant is $Al_2(SO_4)_3 \cdot 18\ H_2O$, which is known as filter alum. The amount of hydrolysis which occurs when filter alum is introduced to the water is a function of the pH of the water, with optimum efficiency achieved at a pH of 7 to 8 (1062).

Existing Literature and Research Presently Under Way

Because chemical coagulation and clarification is probably the most popular water treatment technique, there exists a substantial volume of information on this technology. Table 95 summarizes the available literature located during the study. As shown in the table, most research work has been performed on the elemental group of contaminants as many of the metals are most efficiently removed by chemical precipitation. Substantial study of turbidity removal and virus inactivation/removal have also been conducted. With the current focus on synthetic/organic and biocidal contaminants in water supplies and their potential carcinogenic effects with long-term ingestion, a surge in research in these areas is anticipated.

TABLE 95. LITERATURE REVIEWED PERTAINING TO CHEMICAL COAGULATION AND CLARIFICATION

Contaminants	Reference Number
Water Quality Parameters	
Asbestos	1637, 1638
COD	1243
Color	1243, 1620, 1645
Hardness	1607
Suspended solids	1595
Turbidity	1243, 1588, 1606, 1610, 1619, 1620, 1623, 1665
Elemental Contaminants	
Antimony	1662
Arsenic	520, 531, 1244, 1590, 1596, 1610, 1643, 1655, 1669
Barium	1062, 1590, 1643, 1671
Cadmium	1590, 1610
Chromium	1610
Cobalt	1653, 1654, 1662
Iron	1605, 1610
Manganese	1605, 1610
Mercury	811, 1062, 1610
Molybdenum	1662
Nickel	1662
Selenium	1062, 1610
Vanadium	1662

TABLE 95 (continued)

Contaminants	Reference Number
Biocidal Contaminants	
DDT	1662, 1664
Dieldrin	1662, 1664
Endrin	1662, 1664
Lindane	1664
Parathion	81
Synthetic/Organic Contaminants	
PAH (Polynuclear Aromatic Hydrocarbons)	1627
PCB	708
TTHM (total trihalomethanes)	1676
Biological Contaminants	
Bacteria	25, 103, 219, 380, 1168, 1243, 1382, 1510, 1600, 1601, 1648, 1679
Virus	1168, 1600, 1601

Water Quality Parameters

Asbestos has recently been implicated as a carcinogen of potential danger to workers breathing the fiber in asbestos manufacturing plants. Thus, it is feared that the incidence of asbestos in drinking water supplies may also be a health hazard.

Several studies have investigated the ability of chemical coagulation followed by clarification and/or filtration to remove asbestos from water intended for potable purposes. Lawrence et al. (1638), examined the effectiveness of various methods: straight filtration, diatomaceous earth filtration, chemical coagulation and combinations. The most effective method involved chemical coagulation with iron salts and polyelectrolytes followed by filtration, which resulted in better than 99.8 percent removal from water containing 12×10^6 fibers/ℓ. The optimum ferric chloride dosage was found to range from 6 to 8 mg/ℓ; satisfactory flocs were formed at all test temperatures. Other reported removal efficiencies were reported as follows:

Treatment	% Asbestos Fiber Removal
• Ferric chloride, polyelectrolyte coagulation and filtration	99.8
• Graded sand filtration only	90
• Diatomaceous earth filtration only	96.8
• Bentanite clay, polyelectrolyte coagulation and filtration	99

As a continuation of the above effort, Lawrence and Zimmermann (1637) studied the optimization of alum and polyelectrolyte coagulation for asbestos removal. Optimum removals were obtained with alum concentrations of 30 to 50 mg/ℓ and polyelectrolyte values of 0.3 to 0.6 mg/ℓ. Asbestos removals of over 99.9 percent were achieved. Rapid coagulation and direct filtration was also evaluated and the performance found to be comparable to conventional treatment employing flocculation and sedimentation. In addition, a survey of turbidities and fiber concentrations for several municipal water supplies indicated no correlation between these two parameters.

Shelton and Drewry (1243) evaluated the effectiveness of various coagulants in the removal of COD from a raw surface water supply. Results are summarized in Table 96.

TABLE 96 . A COMPARISON OF THE EFFECTIVENESS OF THE COAGULANTS TESTED ON THE RAW SURFACE WATER

Coagulant and Coagulant Aids	Coagulant con-centration mg/ℓ	Maxi-mum Turbidity Removal percent	Coagulant con-centration mg/ℓ	Maxi-mum COD Removal percent	Optimum Dosage*	
					A	B
$Al_2(SO_4)_3$	15	96.50	20	43.0	12	14
$FeCl_3$	34	94.80	48	73.0	37	32
$Fe_2(SO_4)_3 \cdot nH_2O$	35	90.50	38	88.50	65	42
Cationic flocculant	0.56	46.0	0.51	26	†	1.5
$Al_2(SO_4)_3$ +	8 +		8 +		9.2	7.2
Cationic flocculant	0.25	97.4	2.25	49.5	0.25	0.25
$Al_2(SO_4)_3$ +	10 +		10 +		9	9
Nonionic aid 1	0.25	97.4	0.5	59.3	0.4	0.16
$Al_2(SO_4)_3$ +	10 +		10 +		8.2	‡
Anionic aid 1	0.2	99.65	0.4	97.5	0.4	‡
$Al_2(SO_4)_3$ +	9 +		10 +		7.1	‡
Anionic aid 2	0.2	98.6	0.2	79.5	0.2	‡
$Al_2(SO_4)_3$ +	10 +		7 +		6.4	5.4
Nonionic aid 2	0.2	97.20	0.4	85.00	0.4	0.4
$Al_2(SO_4)_3$ +	10 +		10 +		10	‡
$Na_2OAl_2O_3$	12	98.70	9	86.00	9	‡

* Isoelectric point as indicated by: (A) zeta potential, (B) colloidal titration.

† Isoelectric point was not reached with test dosage.

‡ More than one isoelectric point was indicated.

They concluded that aluminum sulfate was the most effective coagulant for COD removal. Cationic polyelectrolyte additions were ineffective while anionic aids gave better results.

Color is caused by humus, tannins, weeks, algae, soluble wastes, and to a certain extent, metals (1676) In itself, color is not a health hazard; however, it signals the presence of dissolved organics and metals that may be of some health concern. The above researchers evaluated color removal in the same investigation (1243). Ferric chloride was found to perform erratically at differing dosages. Excellent removals were achieved at a dosage of 35 mg/ℓ, but at dosages around 50 mg/ℓ, color was significantly increased. A study by Mangravite et. al (1645) showed that removal of humic acid (one source of color in water supplies) by microflotation produced the same high percentage of removals as the conventional coagulation/sedimentation technique, but at a rate of 5 to 13 times faster. Fulton and Bryant (1620) investigated the use of alum with 18 coagulant aids for color removal. Optimum alum dosages ranged from 20 to 25 mg/ℓ and successfully lowered the color concentration to below five standard units. The use of iron salts, oxidation, adsorption, or polymers alone could not achieve this level of removal.

Hardness removal (CaH and MgH ions predominantly) via time-soda precipitative softening is a common practice at many water treatment facilities. Soft water has generally been considered beneficial in that it reduces the usage of soap and detergents, tastes better, and reduces scaling and precipitation in pipes and water heaters. However, some recent studies, summarized by Cooper (1607) have discovered an apparent inverse relationship between hardness and heart disease; i.e., people drinking soft water showed a higher incidence of heart disease than those using hard water supplies. In light of these results, the overall benefit of the softening process may have to be reevaluated.

Suspended solids removal from water supplies by coated and uncoated diatomite filter was evaluated by Burns et al. (1595) They found that special polyelectrolyte coatings were useful for a final polishing filter process, but that these aids were not as effective as conventional filter aids where large amounts of suspended solids had to be removed.

Turbidity is a uniformly measured water quality parameter specifying the visual clarity of the water. Turbidity is caused by clay and other colloidal matter which in themselves are of no health concern. However, heavy metals, virus, and bacteria may be adsorbed onto the clay particles, and the removal of turbidity can be of importance in regards to potential long-term health effects. Because of its popularity as a quality test, many studies of turbidity removal have been performed as summarized below. Robinson (1665) compared the effectiveness of alum versus polyelectrolytes for removal of turbidity from muddy surface water. Figure 22 summarizes the performance he reports.

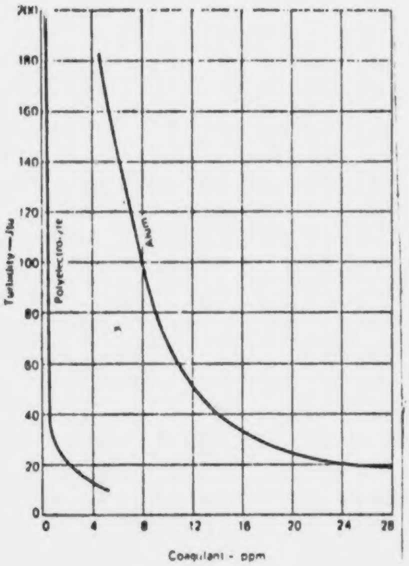

Figure 22. The effectiveness of a small amount of N-607 polymer relative to Alum for raw water with a turbidity of 1-250 Jtu

Fulton and Bryant (1620) investigated turbidity removal at a pilot plant treating water from the Croton watershed of New York City. They found that although alum coagulation (with coagulant aids) could produce water below the 1.0 TU Federal Standard, the American Waterworks Association (AWWA) goal of 0.1 TU could not be achieved without further treatment. Middlebrook et al. (1588) studied the optimum values for operational variables in turbidity removal (alum dosage, flocculation time, and paddle speed).

The following conclusions were drawn from this research:

1. The most significant independent variable was the paddle speed followed by flocculation time and alum dosage.

2. All variables and their first order interactions were significant at the 1 percent level.

3. The optimum alum dosage was 25 mg/ℓ for the conditions under which the study was performed.

4. The optimum paddle speed ranges between 40 and 50 rpm.

5. The optimum flocculation time was approximately 30 min.

6. This study revealed that the significant interactions between the independent variables (alum dosage, flocculation time, and paddle speed) can be utilized in more efficient water treatment. For example, a higher percentage removal of turbidity can be obtained by using low alum dosage and increasing the flocculation time and/or paddle speed. On the other hand, high alum dosages and reduced flocculation times and/or paddle speeds can be utilized.

7. The study shows the importance of the following equipment and methods in the removal of turbidity in water treatment plants:

    a. Variable speed flocculator paddles.

    b. Multiple flocculation basins that can be used in series or parallel operation.

    c. The use of jar tests to determine the optimum alum dosage continuously during the operation of water treatment plants.

Gruninger et al.(1623) compared the performance of ferric chloride versus alum as the primary coagulant and found that the combination of 8 mg/ℓ $FeCL_3$ + 4 mg/ℓ bentonite clay + 0.25 mg/ℓ polymer provided comparable performance to the system with alum and clay; there was approximately 95-98 percent turbidity removal.

Reference 1610 summarizes the performance of various treatment technologies in removing turbidity. These results are -

- Plain sedimentation: 50 to 95 percent, depending totally on the settling characteristics of solids
- Coagulation with sedimentation: 80 to 99 percent
- Rapid sand filtration: 80 to 99 percent with influent turbidity of 10 JTU or less
- Slow sand filtration: 80 to 99 percent with influent turbidity of 10 JTU or less
- Diatomite filtration: 80 to 99 percent with influent turbidity of less than 10 JTU.

Shelton and Drewry (1243) analyzed the effectiveness of different cationic, nonionic, and anionic polyelectrolytes in reducing turbidity. Table 96, presented earlier, displays their results. Aluminum sulfate with anionic coagulant aids was found to be the most effective, achieving 99+ percent removal at a 10 mg/ℓ dosage. Frissora (1619) evaluated the performance of rapid multimedia filtration alone on turbidity removal and achieved removals of 96 to 99.7 percent at high loading rates of 4 to 16 gpm/ft^2. Conley (1606) also studied rapid filtration preceded by chemical coagulation and settling, for turbidity removal at a number of water treatment plants around the country. Table 97 summarizes the results of his study; removals ranged from 80 to 97 percent and averaged 94 percent.

Elemental Contaminants

Many elemental contaminants are readily removed by chemical coagulation and subsequent solids separation steps (settling or filtration). As some of the elemental contaminants are being more closely reviewed for possible long-term health effects, the importance of chemical coagulation as a removal process becomes more pronounced.

TABLE 97. TYPICAL THREE-MEDIA HIGH-RATE FILTRATION PLANT PERFORMANCE

	Turbidity				
Location	Applied JTU	Filtered JTU	Filter Rate gpm/ft^2*	Filter Runs hr	Backwash Water percent
Regina, Sask.	0.5	0.1	4	30	2
Siler City, N.C.	1	0.2	5	30	2
Pasadena, Tex. (industrial)	3	0.5	6	40	1
Corvallis, Ore.	3	0.2	3	50	1
Ocaquan, Va.	1	0.1	4	48	1
Samoa, Ca. (industrial)	10	0.5	5	18	2
Fort Lauderdale, Fla.	1	0.1	4	80	1
Knoxville, Tenn.	1	0.2	5†	60	1
Albany, Ore.	2	0.2	5	20	2
Longview, Tex.	4	0.2	3	48	2
Paris, Tex.	10	0.3	3	50	2
Newport, Ore.	3	0.2	4	20	3
Norristown, Pa.	6	0.3	4	45	1
Springfield, Mo.	1.5	0.2	4½	24	1
Winnetka, Ill.	2	0.3	5	15	2
Stanton, Del.	2	0.2	4	30	3

* Typical summer peak daily flow rate. Hourly peaks are generally in the range of 4-6 gmp/sq ft.

† Under test. Some filters are out of service so that rate could be increased on remaining filters.

Arsenic is one element being closely watched in water supplies as it has a relatively high toxicity, accumulates in the body, and has been associated with the occurrence of cancer (1590). Some 1,056 cases of black-foot disease and skin cancer have been reported in the southwest part of Taiwan. According to statistical data, there is a close relationship between these diseases and the high arsenic content (0.6 to 1.0 mg/ℓ) in deep-well water used for drinking (1669). Since there is no other available water in the area, some practical and economical method to remove the arsenic compounds was urgently needed. To satisfy this need, Shen (1244) performed a lengthy analysis of treatability tests to evaluate the arsenic removal capability of coagulation/settling/filtration processes. Initial coagulant tests showed ferric chloride to be the best chemical, achieving 92 percent As removal at a 30 mg/ℓ dosage. Subsequent testing, however, showed that these removals could be improved by preoxidation before coagulation. Adding 20 mg/ℓ of chlorine and then coagulating with 50 mg/ℓ of ferric chloride provided the best results, achieving 98.7 percent As removal.

Gulledge and O'Connor (520) showed that As removal was affected by pH and doses of suitable coagulants. Arsenic adsorption onto ferric hydroxide exceeded adsorption onto aluminum hydroxide. For both coagulants, increased dosages (up to 50 mg/ℓ) resulted in increased removal. A significant decrease in arsenic removal was seen at pH over 8.0.

Logsdon et al. (1643) studied arsenate removal by precipitation with ferric sulfate and alum, in pilot plant tests. For initial arsenic concentrations of 0.3 mg/ℓ, ferric sulfate treatment yielded 91 to 94 percent removal. Dual media filtration achieved an additional 5 to 7 percent removal for overall removals of 98 to 99 percent and effluent levels of 0.003 to 0.006 mg/ℓ arsenic. Alum treatment was less effective, yielding 75 to 79 percent removal without filtration and 85 to 92 percent removal with filtration (531) Arsenite removals of only 10 to 25 percent were achieved by alum, and 40 to 60 percent by ferric chloride. However, when chlorination preceded coagulant addition, removals equivalent to those reported for arsenate were observed (531). Chlorine apparently oxidized the arsenite to arsenate. For ferric sulfate precipitation, treatment efficiency remained constant from pH 5.5 to 8.5. Above this pH, efficiency declined. With alum, optional precipitation pH was at 7.0, with slight reduction in efficiency at pH below 7.0, and rapid drop-off in removal efficiency at pH above 7.0. Similar pH effects for arsenate treatment by iron and aluminum salts have been observed in another study (1655). Calmon (1596) found that cold lime-soda precipitation with filtration was capable of removing 96.4 percent of the arsenic, whereas ferric sulfate removed less than 50 percent.

The toxicity of antimony is similar to arsenic, although less acute. Very little research has been conducted related to Sb, as other metals appear to be of more importance from a health-effects standpoint. Lime coagulation has been shown to provide Sb removals of better than 90 percent from wastewater effluent (1662). Similar removals may be possible in water supply treatment, but no data were found.

Barium is a toxin that acts as a muscle stimulant. As such, it strongly affects the heart muscles and causes high blood pressure and nerve blockage (1590). Barium removal - only for wastewater treatment - was reported by several researchers. Sigworth and Smith (1671) found during pilot wastewater plant tests that lime coagulation and settling was an effective method, resulting in a 99 percent Ba removal. Dosage rates were 45 mg/ℓ $FeCl_3$, 260 mg/ℓ lime plus 20 mg/ℓ $FeCl_3$ for the low lime test, and 600 mg/ℓ lime for the high lime run. Logsdon (1643) found improved removals with the lime softening process. Treatment efficiency was pH dependent, increasing to a maximum of 98 percent at pH 10.5, and dropping off at higher pH values. Barium is not appreciably removed by ferric or alum coagulation.

Cadmium has a high toxic potential, accumulates in various organs of the body, and is a causative factor in high blood pressure (1610). Only minute traces of cadmium are found in natural waters. However, several industries discharge cadmium in their wastewaters, including metallurgical alloying, ceramics manufacture, electroplating, inorganic pigments, textile printing, chemical industries, and acid mine drainage. Very little data are available regarding the use of coagulation for Cd removal at water treatment plants. The Summary Reference (1610) lists removals at 50 to 90 percent with coagulation followed by filtration. Unpublished data developed by EPA indicates that lime softening and ferric sulphate coagulation above pH 8 achieves Cd removals of over 98 percent.

Chromium in the hexavalent form is highly toxic to man and has been shown to be carcinogenic to man when inhaled. Volkert and Associates, in a study for EPA (1610) summarized that coagulation and sedimentation with filtration is capable of removing 50 to 90 percent of the insoluble chromium (Cr) forms present in water supplies.

The hexavalent form is much more difficult to remove with typical water treatment coagulants than is the trivalent form. It is desirable, therefore, to reduce the chromium present to trivalent prior to treatment. Laboratory studies by EPA (unpublished) indicate that trivalent chromium is removed at better than 95 percent levels by ferric sulphate up to pH 9.5, removed at 90 percent levels by alum up to pH 8.5, and removed at 98 percent level by lime softening at pH from 10 to 11.5. Hexavalent chromium, however, is not effectively removed by any of these coagulants. Cobalt (Co) concentration in potable water sources seldom reaches levels that require special consideration. Cobalt is currently best treated by lime coagulation, although as much as 2 mg/ℓ may be left in the effluent (1662). Nilsson (1654) found that lime coagulation was capable of achieving 88 percent removal of cobalt. Another study (1653) found that cobalt precipitation could be enhanced by the addition of chitosan or a chelate as a polishing agent. An EPA study (1662) concluded that similar precipitation techniques are also applicable to molybdenum (Mo), nickel (Ni), and vanadium (V) removal.

Iron and manganese, often found together, are important constituents of potable water supplies because they can import unwanted aesthetic qualities. From a health standpoint, however, iron and manganese do not pose any significant threat in the concentrations normally found in surface and groundwater supplies. Therefore, removal of these constituents will not be discussed in any depth here. Effective removal can be provided by aeration or more commonly, by chemical precipitation, sedimentation, and filtration (1605).

Recent attention has been focused on the contamination of water supplies by organic and inorganic mercury (Hg) by industrial discharges. Fortunately, recent rests by the EPA of 273 water supplies across the country showed very low mercury levels in nearly all of them. Even so, the performance of conventional water treatment technologies and new techniques for removing Hg from water is of interest. Logsdon and Symons (811) investigated the efficiency of conventional water treatment processes in removing Hg. Bench scale studies of chemical coagulation and settling yielded the following conclusions:

1. Mercury removal during coagulation was related mainly to adsorption of mercury onto the turbidity-causing agents in the water.

2. In raw waters with low turbidity, ferric sulfate was more effective than alum for removal of inorganic mercury. Removals ranged from 40 to 60 percent over a turbidity range of 2 to 100 JTU's.

3. Removal by coagulation or adsorption onto turbidity-causing agents was less for methyl mercury than for inorganic mercury.

4. Inorganic mercury removal by softening was most effective in the pH range of 10.7 to 11.4 and is thought to be related to adsorption onto magnesium hydroxide floc.

5. Methyl mercury was not removed by softening.

The authors concluded by stating that as long as environmental levels of mercury in raw water remain low (near drinking water standards), extremely high removals will not be required, and conventional technology should be sufficient.

Molybdenum can occur naturally in drinking water supplies at concentrations up to 20 μg/ℓ due to weathering of minerals. It can also be found in water as a result of waste discharges from industries related to the manufacture of glassware, ceramics, printing inks, electrical equipment, and certain steel alloys. Virtually no data relative to molybdenum removal from water supplies are available, because molybdenum is not currently seen as a primary health hazard.

Nickel occurs naturally at concentrations up to 0.072 mg/ℓ with an average concentration of about 0.005 mg/ℓ. Wastewater discharges from industries associated with nickel-plating, nickel alloying, storage battery manufacturing, organic hydrogenation, and the manufacture of nickel-chrome resistance wire can contribute to the presence of nickel in water sources (43). In large doses, Ni can be harmful; however, in this country, natural concentrations are low, and lime coagulation appears to be successful in removing large percentages of Ni from wastewater. No literature pertinent to Ni removal by water treatment facilities was found.

Alum and ferric sulfate coagulation and lime softening are only moderately effective for selenite (Se^{+4}) removal, and are largely ineffective for selenate (Se^{+6}) removal (1643). Studies with ferric sulfate (30 mg/ℓ) at a pH of 5.5 yielded removals of 85 percent from river water containing 0.03 mgSe^{+4}/ℓ. Removals decreased as the pH increased. The maximum achievable Se^{+4} removal with alum (100 mg/ℓ, pH 6.9) was 32 percent. A maximum removal of 45 percent was obtained with lime softening. No conventional method achieved greater than 10 percent removal of Se^{+6}.

No data were located relative to vanadium removal from water supplies. In wastewater treatment, however, very good vanadium removals were achieved via coagulation with iron or aluminum salts or chitosan polymers (1662).

Biocidal Contaminants

Recently, there has been increased awareness of the potential health hazards of biocidals. Limited data, however, are available in the literature regarding the removal of these constituents from water supplies by chemical coagulation: Robeck et al. (1664) studied the effects of various water treatment processes on pesticides. Their results, in regards to pesticide removal via chemical coagulation (alum) and filtration, are shown in Table 98 below.

TABLE 98. PERCENTAGE OF PESTICIDE REMOVED BY CONVENTIONAL WATER TREATMENT

Pesticide	Coagulation and Filtration			
	Load - ppb			
	1	5	10	25
Lindane	<10	<10	<10	
Dieldrin	55	55	55	
DDT			98	97
Parathion			20	
2, 4, 5-T ester			63	
Endrin	35		35	

* Total hardness as $CaCO_3$ reduced from 260 to 33 ppm and pH increased from 7.6 to 10.4.

As can be seen, DDT was readily removed, whereas lindane, parathion, and endrin were not. Dieldrin and 2, 4, 5-T ester are removed at slightly better than 50 percent. Softening with lime and soda ash and with an iron salt as a coagulant did not improve on the removals obtained with alum coagulation alone.

Another study (1640) evaluated the effect of $KMnO_4$ as an oxidant and precipitating agent. It was found that $KMnO_4$ was not significantly effective in removing lindane, but was capable of removing over 80 percent of the heptachlor present in under 5 hours. Removals of DDT were less than 20 percent in 48 hours, and endrin was totally unaffected by the process.

Synthetic/Organic Contaminants

In recent years, concern has been expressed over the possible occurrence of certain carcinogenic compounds in drinking water (World Health Organization, 1964). A group of compounds which has received particular attention is the polynuclear (polycyclic) aromatic hydrocarbons (PAH), some of which are potent carcinogens under certain conditions. It is, however, far from certain that these compounds are significant when present in the trace amounts found in drinking water. Clearly, further research is needed into both the levels and health effects of PAH in the environment (1627).

Harrison et al. conclude (1627): "The reliability of much of the information concerning the removal of PAH by conventional water treatment processes is open to considerable doubt. Field experiments have frequently ignored retention times within the works, and hence been rendered unreliable at the sampling stage. Laboratory studies have commonly used unrealistic high concentrations of PAH, and the high removals achieved may be largely explained often by the low solubility of the PAH themselves. Further analytical work is required in this field, and fundamental studies of the chemical changes that occur upon chlorination of these compounds at low concentrations are necessary. Increasing water reuse makes the need for this type of information particularly acute."

After the EPA disclosed its findings of organic chemicals in New Orleans' water, it was decided to conduct another, more inclusive study at specific sites across the U.S. In this study, Symons et al. (1676) found that precipitative softening at water treatment plants increased the concentration of trihalomethane in the product water from an average of 0.49 µm/ℓ (80 locations) to 0.84 µm/ℓ (17 plants with softening). This indicates that chlorination at a higher pH will produce higher concentrations of trihalomethanes.

Kinoshita and Sunada (708) performed an experiment on the removal of PCB by chemical coagulation. Bentonite, sodium carbonates, and aluminum sulfate were added to a solution of 100 ppb PCB. After settling, the PCB concentration has been reduced 90 percent, suggesting that conventional water treatment by coagulation and settling or filtration provides a margin of safety against the ingestion of PCB's in raw water supplies.

Biological Contaminants

The use of diatomite filtration with and without chemical addition is capable of removing bacteria from water supplies, McIndoe (1648) summarized research in this area as follows:

- Work by the U.S. Army Engineering Research and Development Laboratories at Fort Belvoir, Virginia, showed that a diatomite filter using a coarse grade of filter aid was capable not only of 100 percent removal of the entamoeba cysts, but also of partial removal of other pathogenic organisms. Work done at Rutgers University has shown that even the coarsest grade of diatomite gave coliform removals of greater than 50 percent for influent levels of 210 to 1300/100 mℓ. Finer grades gave "complete" coliform removals of influent levels up to several thousand per 100 mℓ.

- Work by S. L. Chang (1600, 1601) and J. V. Hunter (617) with solutions of iron salts added to bacteria-contaminated water has indicated a reaction takes place between the iron ions and the protein sheath of both bacterial and viral organisms. Subsequent removal of the iron by the diatomite MgO process has produced total count reductions on the order of 99 percent, with fecal strep and Escherichia coli counts of zero.

- Waters so clarified would be expected to be readily disinfected by usual practices.

Amirhor and Englebrecht (25) also analyzed the potential use of uncoated and polyelectrolyte-aided diatomaceous earth DE filtration for bacterial virus removal. They concluded that all uncoated DE filtration was not effective for virus removal, but that precoating of cationic polyelectrolyte greatly improved removals. Certain variables such as pH, level and concentration of polyelectrolyte coating, and virus concentration affected removals.

Wolf et al.(1510)conducted a large-scale pilot study of virus removal by both lime and alum. They demonstrated that virus removals from secondary effluents by alum coagulation-sedimentation and coagulation-sedimentation-filtration processes are essentially the same as described in the literature using smaller-scale processes. Removals of bacterial virus as high as 99.845 percent for coagulation-sedimentation and 99.985 percent for coagulation-sedimentation-filtration processes were observed at an Al:P ratio of 7:1.

At a lower alum dose there was a marked decrease in virus removals. At an Al:P ratio of 0.44:1, removals of only 46 percent of f_2 coliphage and 63 percent of polio virus by the coagulation-sedimentation process per se were observed.

High lime treatment of secondary effluents achieved very high degrees of virus removal, but the percentage has not yet been quantified.

Englebrecht and Chaudhuri (219) conducted a study to extend the knowledge in this field. Their conclusions follow:

1. Chemical coagulation and flocculation is an effective process in removing viruses from water. Removals in the range of 98.0 to 99.9 percent can be expected.
2. The presence of bivalent cations like calcium and magnesium up to a concentration of 50 mg/ℓ each does not interfere with the efficiency of the process.
3. The efficiency of virus removal is reduced when the raw water contains organic matter.
4. Intelligent use of commercially available cationic polyelectrolytes with or without hydrolyzed metal ions may markedly increase the efficiency of the coagulation and flocculation process in removing virus.
5. Virus particles remain active in the settled sludge following their removal from water by coagulation and flocculation, and can be recovered from the floc by various eluants. Therefore, proper care should be taken in sludge disposal.

Thorup et al. (1382) conducted a study to determine the effectiveness of polyelectrolytes used as coagulant aids for the removal of virus from artificially seeded water. They found that the cationic polyelectrolyte performed more acceptably than the anionic and nonionic types and was beneficial in aiding floc formation under conditions of poor coagulation. However, the 80 to 94 percent removal achieved under these circumstances was well below the 99^+ percent usually considered acceptable. In instances of adequate coagulation, however, cationic polyelectrolytes did not increase virus removal beyond the levels obtained with unaided $Al_2(SO_4)_3$ or $FE_2(SO_4)_3$ coagulation.

Shelton and Drewry (1243) performed a literature search of virus (f2 bacteriophage) removal via coagulation and, based on the results, studied the effectiveness of different chemical coagulants and polyelectrolytes for this purpose. Their conclusions were:

1. Aluminum sulfate, ferric chloride, and ferric sulfate are the most promising of the group of primary coagulants tested, from the standpoint of f2 bacteriophage removal, all producing greater than 99 percent removal. Aluminum sulfate is considered to be the most effective coagulant, whereas ferric chloride and ferric sulfate are comparable and less effective.

2. The cationic polyelectrolyte tested is not satisfactory as a primary coagulant because of its poor floc formation and settling characteristics. Virus removals with the cationic flocculant were only moderate (92 percent).

3. Anionic, nonionic, and cationic polyelectrolytes have only minor significance in virus removal. It is noted, however, that the anionic coagulant aids are useful in widening the effective dosage range for good virus reduction. The nonionic coagulant aids give moderately better results, which are attributable to an ability to form more dense and "sticky" floc. This improvement of virus removal is indirect, because the nonionic polyelectrolytes do not form a virus-ion complex with the virus protein coat. The cationic coagulant aid produces a moderate improvement in virus removal; however, it is questionable if this improvement would be economically justifiable for a full-scale operation.

4. The use of sodium aluminate with aluminum sulfate does cause a marked increase in virus removal, to 99.9 percent overall. This increase, however, could be used only for "special-case" situations, since the sodium aluminate dosage for optimum virus removal is not compatible (by a factor of 3) with turbidity and COD optimums.

Thayer and Sproul (1679) of the University of Maine did extensive research concerning the effects of water softening upon virus inactivation. Their primary objective was the determination of the effect of chemical precipitation in water softening upon bacterial viruses. Inactivation of T2 virus varied widely, with the best results, on the order of 99.999 percent inactivation, occurring when magnesium hydroxide was the only precipitate formed. The standard excess lime-soda-ash process also produced good results with about 99.45 percent virus reduction.

Lime flocculation with rapid sand filtration, a long-used standard water-treatment method, was investigated for its virus-removal characteristics by Berg et al. (103) at the Cincinnati Water Research Lab. Conclusions based on their study using polio virus I (LSC nacuine) exclusively were that (1) flocculation of secondary effluent yielded up to 99.86 percent removal of the virus, dependent upon the coagulant dose and the pH level attained, and (2) the total removal by lime flocculation and rapid sand filtration through an 8-in-deep filter achieved a maximum of 99.997 percent.

The ASCE Journal of Sanitary Engineering published an article in 1961 summarizing the virus work done up to that time (380). Table 99 shows the results of this work. As shown, the dosage of flocculant alone is not a measure of the efficiency of the process.

Robeck et al. (1168) investigated the fate of human viruses in rapid filtration processes with and without flocculation and settling. They found that if low alum doses were fed just ahead of the dual-media filter (operating at 6 or 2 gpm/ft^2), more than 98 percent of the virus was removed. When the alum dose was increased and conventional flocculators and settling added, removals were increased to over 99 percent. When turbid water was treated, a floc breakthrough was usually accompanied by an increase in virus penetration. Polyelectrolyte doses as low as 0.05 mg/ℓ increased floc strength and helped to prevent such breakthroughs.

The current state of knowledge indicates that chemical flocculation, settling, and filtration are effective in removing virus from water. Removals of 99$^+$ percent have been reported under proper operating conditions. However, more research is still needed in the area to fully determine the most effective doses of coagulants and coagulant aids, the physiochemical effects of turbidity, pH, temperature, and colloidal charge, and to develop optimal operating parameters.

TABLE 99. REDUCTION OF HUMAN ENTERIC VIRUSES IN WATER BY CHEMICAL FLOCCULATION

		Flocculation Process			
Virus	Type of Water	Stages	Coagulant Added[a] ppm	Floc Amount or Description	Virus Removal
Polio	Tap	1	100		Little[b]
	Raw	1	100		Some[c]
Polio	River	1	136		Some to significant[d]
		1	410		Some
		1	136-273	0.4-1.03 mi per l	Little to some
		1	273-546	1.5-2.2 mi per l	Significant
Infectious hepatitis[e]	Distilled	1	69		Little to some
Coxsackie A5	Spring	1	28-45		Some
Coxsackie A2 (purified)	Ohio River (16-255ppm turbidity)	1	15	Good	95.7% at 25°C
		1	25	Very good	95.9% at 5°C
		2	15	Very good	98.6% at 25°C
			15*	Very good	99.8% at 25°C
		2	25	Very good	99.6% at 5°C
			25*	Very good	99.9% at 25°C

[a] Alum except where marked with an asterisk; asterisks indicate ferric chloride.

[b] Probably less than 25%.

[c] Probably less than 50%.

[d] More than 50%.

[e] Gauze-strained fecal suspension in distilled water.

Chang et al. (1601) performed comprehensive studies of the dynamics of removal of bacterial virus by aluminum sulfate flocculation. From their observations they concluded:

1. Flocculation by aluminum sulfate can remove high percentages of virus, and within the zone of flocculation, higher doses produced greater efficiency.
2. The virus is concentrated in the floc sediment and is not destroyed, but only temporarily inactivated. It will become active again, if dissociated from the aluminum.
3. Virus inactivation is believed to be the result of the formation of an aluminum protein salt in the virus.

Chang et al. performed another study (1600) evaluating the efficiencies of alum and ferric chloride in removing coxsackie and bacterial viruses. They analyzed the effects of coagulant dosage, pH, and rate and method of stirring on virus removal. A 40 mg/ℓ dose of $Al_2(SO_4)_3$ yielded an 86.3 percent removal of coxsackie virus and a 93.5 percent removal of bacterial virus. Under similar testing conditions, 20 and 40 mg/ℓ of $FeCl_3$ facilitated 96.6 and 99.3, and 98.1 and 99.9 percent removal of these viruses, respectively.

DISINFECTION

Introduction

Disinfection refers to the inactivation or destruction of pathogenic microorganisms. Disinfectants (chlorine, ozone, ultraviolet and ionizing radiation) also have secondary applications, particularly as oxidants for the removal of organic contaminants. Both applications are included in the literature pertaining to the treatment of drinking water with disinfectants (Table 100).

In the United States, the traditional disinfectant is chlorine, which is usually applied to the water in gaseous form as the last treatment step. The chlorine gas (Cl_2) reacts with the water to form hypochlorous acid (HOCl), which dissociates at pH values above 7 to form the hypochlorite ion (OCl^-) (651). Occasionally hypochlorous acid, hypochlorite, or chlorine dioxide (ClO_2) are used instead of chlorine gas.

After chlorine, the most popular disinfectant is ozone, especially in Europe. Ozone (O_3) is an unstable gas and must be generated at the site and used immediately. It is a powerful oxidant. Both its germicidal and oxidizing properties seem to be the result of the formation of several free radicals in water (HO_2, OH, HO_3^+), which will attack almost all organic compounds.

Ultraviolet and ionizing radiation have been used in pilot plant and small industrial applications. Like ozone, they seem to act by forming a series of free radicals in water, which can attack organic bonds. Ultraviolet is incapable of acting at more than a few centimeters depth, and both forms of radiation are highly susceptible to interference from turbidity and suspended matter. Ionizing radiation requires radioisotopes and the concommitant shielding and elaborate safety precautions.

All of the disinfectants have disadvantages that prevent any of them from being universally applicable. For a given situation the choice depends largely on the water quality, types of microorganisms in the water, desirability of nondisinfection applications, and cost.

TABLE 100. LITERATURE REVIEWED PERTAINING TO WATER DISINFECTION

Contaminant	Reference Number		
	Chlorination	Ozonation	Radiochemical
Water Quality Parameters			
Ammonia	651, 1635, 1675, 1685	651	980
BOD, COD, TOC	651, 1667, 1675	651, 707, 1429, 1651	
Nitrites		651	
Elemental Contaminants			
Iron		818, 1610, 1651	
Manganese		707, 1610, 1651	
Biocidal Contaminants			
Chlorinated hydrocarbons	1594, 1610	1594, 1610	1610
Organo-phosphorus			1610
Carbamates			1610
Synthetic/Organic Contaminants			
PCBs			708, 1610
General	1627, 1632, 1667, 1675	1610, 1627, 1651	1610
Biological Contaminants			
Viruses, general	101, 214, 257, 468, 651, 815, 970, 1168, 1291, 1627, 1675	214, 468, 651, 818, 970, 1291, 1429, 1651	
Coliforms	214, 1635, 1662, 1667	113, 214, 542, 651, 1662	
E. Coli	101, 651, 707, 1429	707, 1429, 1651	214, 980, 1414
Clostridium		1429	
Protozoa	214, 1429	214, 651, 1429	

Water Quality Parameters

Although disinfectants are seldom specifically employed to remove any of the water quality parameters, there are certain chemical reactions that make the idea feasible. The reactions of chlorine with ammonia are the best known. If allowed to go to completion, the ammonia will be oxidized to nitrogen gas and will thus be removed from the water. Ozone does not react with ammonia (651). Ammonia is seldom an issue, since it is not a common constituent of most water supplies.

The organics, as represented by BOD, COD, and total organic carbon, in drinking water are susceptible to oxidation by disinfectants. However, reactions other than oxidation may produce potentially hazardous compounds. For instance, Rook (1667) and McClanahan (651) reported that chlorine reacted with humic and sulvic substances, forming chlorinated organic compounds. These chlorinated compounds are much more resistant than the precursor compounds to both biodegradation and chemical oxidation. Consequently they persist in a water supply that is not treated any further than chlorination. Some of the chlorinated compounds formed are suspected to have carcinogenic properties.

Ozone is even more reactive toward organic compounds than is chlorine. With ozone, though, the reactions are almost exclusively oxidation, with few if any hazardous compounds formed in side reactions. The formation of ozonides and similar compounds has been postulated, but there has been no evidence to data demonstrating their formation during the ozonation of water. Morris (1651) reported an increase in BOD after ozonation and attributed it to the breakdown of nonbiodegradable organic molecules into simple, degradable compounds.

Murphy (980) recently demonstrated that gamma radiation had an oxidizing effect on organic compounds similar to that of ozone. Ultraviolet radiation theoretically has a similar effect.

It should be noted that the use of chemical disinfectants for oxidizing nonbiological contaminants interferes with the primary disinfection role of these chemicals by consuming the disinfectants. To achieve proper disinfection in high organic waters, for instance, requires large increases in applied dosages. Some water-borne disease outbreaks are attributed to improper disinfection of highly organic water supplies.

Elemental Contaminants

In general, the disinfectants have no effect on elemental concentrations. Elements would have to be in a reduced state before oxidizing disinfectants could have an impact. This is not likely in most drinking water supplies. Exceptions are iron and manganese, which are more soluble in their lower oxidation states. Morris (1651) reported that ozone readily oxidized soluble iron and manganese to the insoluble oxides, which could then be removed by sedimentation or filtration.

Trivalent chromium can be oxidized to the hexavalent form, which is more toxic and difficult to remove with conventional coagulation/filtration processes.

Biocidal Contaminants

Chlorinated hydrocarbon biocides are generally resistant to chemical oxidations. Stone et al. (1662) reported that chlorine was not a particularly effective oxidant for such biocides. Ozone was more effective, but removal efficiencies varied widely from 16 to 93 percent, depending on the type of biocide, ozone concentration, and contact time (Table 101). Stone et al. reported other ozone studies that yielded 50 percent removal of endrin, 75 percent removal of lindane, and approximately 100 percent removal of dieldrin and alrin. They also stated that ultraviolet radiation could completely eliminate carbamate biocides, reduce aldrin by 45 percent, and reduce endrin and dieldrin by 18 percent.

Buescher et al. (1594) studied the chemical oxidation of chlorinated hydrocarbons in water. They included the following:

- Lindane concentrations in aqueous solutions were readily decreased by ozonation and only partially affected by potassium permanganate. Treatment with chlorination, peroxides, and aeration had no measurable effects.

- Aldrin in aqueous solutions was readily attacked by chlorination, potassium permanganate, ozone and aeration; peroxides had no measurable effects.

- Dieldrin concentrations in aqueous solutions were decreased by ozonation and aeration.

- Lindane and aldrin in aqueous solution were found to be volatile. The degree of volatility may be an indication of the susceptability of that pesticide to chemical treatment.

- Lindane in the presence of other naturally occurring trace organics found in river water was readily attacked by ozonation, though aeration had only a minor effect.

TABLE 101. EFFECT OF OZONATION ON CHLORINATED HYDROCARBON INSECTICIDES (1610)

		γ-BHC		Dieldrin		DDT		TDE (DDD)	
Time of Ozonation	Ozone Absorbed mg/ℓ	Before Ozonation	After Ozonation	Before Ozonation	After Ozonation	Before Ozonation	After Ozonation	Before Ozonation	After Ozonation
					μg/ℓ				
5	8.8	1.32	0.88	1.29	1.08				
10	18.3	1.39	0.81	1.30	0.66				
20	36.0	1.31	0.34	1.31	0.22				
5	11.7					2.00	0.54	2.00	0.62
10	20.0					2.00	0.46	2.00	0.43
20	38.2					2.00	0.14	2.00	0.13

Synthetic/Organic Contaminants

As previously mentioned under "Water Quality Parameters," both chlorine and ozone readily react with dissolved organics. However, synthetic organics are often more resistant to oxidation than the natural organics. Rosenblatt (651) indicated that chlorine reacted with many organics or give both chlorinated and oxidation products, but that there was no reaction with many others (Table 102). Ozone is an effective oxidant against the phenobics and organic nitrogen compounds, but not against many of the simpler organic molecules, such as ethanol. Harrison et al. (1627) reported that chlorine was more effective than ozone against benzo(a) pyrene.

Many of these synthetic organics, such as nitrobenzene, benzo(a) pyrene, oniline, and ethyl benzene are reportedly carcinogenic. While these chemicals are seldom found in drinking water supplies at concentrations exceeding a few ppm, the postulated no-threshold-dose character of many carcinogens makes even one molecule a potential hazard. Note also that chlorine is suspected of producing chlorinated organic compounds which may themselves be carcinogenic.

Kinoshita and Sunada (708) investigated the effects of irradation on PCB in water. They concluded that PCB in aqueous microparticulate colloidal solution is destroyed by ionizing irradation (up to 95 percent), but that its resistance to radiation is far greater than other chlorinated hydrocarbons used as penthachlorophenol or DDT, and other pesticides such as parathion. They also found that the acute toxicity of the irradiated PCB solution was far less than the nonirradiated solution for striped shrimps.

Biological Contaminants

The major application of disinfectants is against biological contaminants. In this light, the disinfectants have been primarily on their effectiveness in controlling biologicals (e.g., bacteria, viruses, protozoa, parasitic worms).

Chlorine is the traditional disinfectant in the United States. It is effective to some extent against all types of pathogenic organisms found in water. Bacterial kills of at least 99 percent are considered normal (1662), and 4 to 5 log reductions are not unusual. Both Sobsey (1291) and Long (815) reported virus reductions of up to 99.99 percent. Reference 257 summarized several researches on virus destruction by chlorine as shown in Table 103. Chlorine can be very effective against free-swimming protozoa and parasitic worms. Chlorine has the further advantage of persistence; given a sufficiently large dose, a low residual chlorine concentration will remain in the water after treatment, providing continued

TABLE 102. PROBABLE REACTION PRODUCTS OF CHLORINE AND SOME TYPICAL ORGANIC COMPOUNDS FOUND IN POLLUTED WATER SUPPLIES (651)

Organic Compound	Probable Reaction Products
Alcohols	
Methanol	None
Isopropanol	None
tert-Batanol	None
Ketones	
Acetone	None
Benzene and Derivatives	
Benzene	None
Toluene	None
Ethylbenzene	None
Benzoic acid	None
Phenol and Phenolics	
Phenol	Moro-, di-, and trichlorophenols; non-aromatic oxidation products
m-Cresol	Mono-, di-, and trichlorocresols; non-aromatic oxidation products
Hydroquinone	p-Benzoquinone, non-aromatic oxidation products
Organic Nitrogen Compounds	
Aniline	Mono-, di-, and trichloroanilines; non-aromatic oxidation products
Dimethylamine	N-Chlorodimethylamine, oxidation products
Nitrobenzene	None

TABLE 103. VIRICIDAL EFFICIENCY OF FREE CHLORINE IN WATER (257)

Investigator	Virus	Temp. °C	Final pH	Free Chlorine mg per ℓ	Virus Destruction
Chang et al.	Part. purif. Theiler's virus in tap water	25-27	6.5-7.0	4.0-6.0	98.6% in 10 min
		25-27	6.5-7.0	4.0-6.0	99% in 5 min
Neefe et al.	Feces-borne Inf. Hepat. virus in dist. water	Room	6.7-6.8	3.25	30 min cont. time protected all of 12 volunteers
Lensen et al.	Purif. Polio II in dist. and lake water	19-25	7.4-7.9	1.0-1.5	10 min cont. time protected all of 164 inoc. mice
Clarke and Kabler	Purif. Coxsackie A2 in demand-free water	3-6	6.9-7.1	0.58-0.62	99.6% in 10 min
		3-6	6.8-7.1	1.9-2.2	99.6% in 4 min
		3-6	6.9-7.1	3.8-4.2	99.6% in 2½ min
		3-6	8.8-9.0	1.9-2.0	99.6% in 24 min
		3-6	8.8-9.0	3.7-4.3	99.6% in 9 min
		3-6	8.8-9.0	7.4-8.3	99.6% in 5 min
		27-29	6.9-7.1	0.16-0.18	99.6% in 4 min
		27-29	6.9-7.1	0.44-0.58	99.6% in 3 min
		27-29	8.8-9.0	0.10-0.18	99.6% in 10 min
		27-29	8.8-9.0	0.27-0.32	99.6% in 7 min
		27-29	8.8-9.0	0.92-1.0	99.6% in 3 min
Weidenkopf	Purif. Polio I (Mahoney) in demand-free water	0	6.0	0.39	99.6% in 3½ min
		0	6.0	0.80	99.6% in 1½ min
		0	7.0	0.23	99.6% in 8 min
		0	7.0	0.53	99.6% in 4½ min
		0	8.5	0.53	99.6% in 16 min
		0	8.5	1.95	99.6% in 7½ min
		0	8.5	5.00	99.6% in 3 min
Kelly and Sanderson	Purif. Polio I (Mahoney) in demand-free water	25-28	7.0	0.21-0.30	99.9% in 3 min
		25-28	9.0	0.21-0.30	99.9% in 8 min
	Purif. Polio III (Saukett) in demand-free water	25-28	7.0	0.11-0.20	99.9% in 2 min
		25-28	9.0	0.11-0.20	99.9% in 16 min

TABLE 103 (continued)

Investigator	Virus	Temp. °C	Final pH	Free Chlorine mg per ℓ	Virus Destruction
Kelly and Sanderson (cont'd)	Purif. Coxsackie B5 in demand-free water	25-28	7.0	0.21-0.30	99.9% in 1 min
		25-28	9.0	0.21-0.30	99.9% in 8 min
		1-5	7.0	0.21-0.30	99.9% in 16 min
		1-5	8.0	0.21-0.30	99.9% in 30 min
Clarke et al.	Purif. Adenovirus 3 in demand-free water	25	8.8-9.0	0.20	99.8% in 40 sec-50 sec
		25	6.9-7.1	0.20	99.8% in 8 sec-16 sec
		4	8.8-9.0	0.20	99.8% in 80 sec-100 sec
		4	6.9-7.1	0.20	99.8% in 8 sec-10 sec

germicidal action. This prevents regrowth and protects against accidental contamination during distribution.

Despite chlorine's usefulness, it does have some serious disadvantages. Clarke et al. (240) and Sobsey (1291) reported the isolation of viruses in chlorinated drinking water in Paris (1 pfu/300ℓ) and South Africa (1 pfu/10ℓ). In view of the fact that only one to two viruses of some types are sufficient to cause infection, anything less than 100 percent inactivation may be unacceptable. But with chlorine even the absence of any living viruses still may not be acceptable. McClanahan (1646) reported that chlorine removes the protein coat of a virus - thus rendering the virus technically nonviable - but may leave the infectious nucleic acid core intact. Consequently, a water supply free of any living viruses may still be infectious.

Chlorine is also relatively ineffective against protozoal and helminthic cysts. These encysted organisms are very resistant to oxidation by chlorine.

Ozone is equally as effective as chlorine against bacteria and viruses and has a much faster reaction rate. Once the proper "threshold" ozone dose is applied (usually less than 5 mg/ℓ), the bactericidal action is almost instantaneous. Tests have shown ozone to be between 600 and 3,000 times more rapid than chlorine in its destruction of bacteria (1646). McClanahan (651) was unable to recover viable nucleic acids from ozonated water, suggesting that virus destruction was complete, as opposed to the action of chlorine. Venosa (1429) reported that protozoal cysts resistant to chlorine were easily inactivated by ozone. Futhermore, the biocidal character of ozone is not affected by pH, as is the biocidal character of chlorine.

Ozone is not without problems; however, ozone leaves no residual. It has a fairly short half-life in water and rapidly loses all disinfectant ability. Experience in Europe has revealed few problems along these lines, but the added margin of safety with chlorine has worked against the adoption of ozone in the United States. The versatility of ozone and its relatively greater disinfecting ability has led to the suggestion that initial ozonation could be followed by low-level chlorination to provide a residual. However, little research has been conducted along these lines.

Murphy (980) and Vajdic (1414) both reported that gamma radiation was as good a disinfectant as chlorine against bacteria and was somewhat better against the more chlorine-resistant biologicals. Ultraviolet is a proven bactericide, but research on other biocidal characteristics has been limited. Nevertheless, any radiation treatment suffers from operational difficulties, and, like ozone, provides no residuals.

The sanitation districts of Los Angeles County are in the final stages of an extensive study for EPA and the California State Water Resources Board. This study titled "The Pomona Virus Study," has evaluated virus removals by various combinations of tertiary treatment processes and followed by chlorination or ozonation. Their conclusions in part are that the majority of virus inactivation occurs during disinfection and the main function of the preceeding tertiary unit processes was that of removing substances (turbidity, organics, etc.) which interfere with efficient disinfection. In virus seeding experiments involving combined chlorine residuals of 5 mg/ℓ average seed virus removals of 4.7 to 5.1 logs were achieved. With 10 mg/ℓ of combined chlorine residuals, 5.2 logs of virus removal were achieved. In seeding experiments involving ozonation, average log virus removals ranged from 5.1 to 5.4 logs.

ADSORPTION ONTO ACTIVATED CARBON AND OTHER MATERIALS

Introduction

Activated carbon adsorption (or simply carbon adsorption) is employed to remove color, odor, taste, and refractory organic compounds from water. Many water treatment plants presently pass their effluent through a carbon column or fine-grain carbon bed to polish the final product. Available data indicate that carbon adsorption is an effective method for removing synthetic and natural organic contaminants, particularly chlorinated hydrocarbons and organophosphorus pesticides, from plain water. Carbon adsorption may also be used to remove some metals. There is some natural adsorption, but metal removal can be greatly enhanced by the addition of an organic chelating agent prior to passage through the carbon. The carbon will readily adsorb the chelating agent, thereby also removing the complexed metal.

The literature that has been reviewed on the effectiveness of the adsorption process in potable water treatment is summarized in Table 104. The major portion of the research to date has involved tertiary wastewater treatment application, although there has been substantial work also in the water treatment field shown in Table 104. With the recent concern over residual organics in U.S. water supplies, one can anticipate increased activity in both pilot and full-scale activated carbon systems for treating water supplies.

In addition to activated carbon, synthetic polymeric adsorbents have been extensively tested and show promise for potable water treatment. They are not widely used in water treatment plants but have been tested in pilot-scale installations. Some tests have indicated higher removal efficiencies for synthetic adsorbents than for activated carbon for some contaminants. Inorganic adsorbents, such as clays and magnetite, are also capable of contaminant removal.

Water Quality Parameters

Activated carbon adsorption was used following chemical coagulation and rapid sand filtration at the much-publicized Windhoek, South Africa, water treatment plant. The influent

TABLE 104. LITERATURE REVIEWED PERTAINING TO ADSORPTION

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	1649, 1674
BOD	1660, 1674
COD	1660
Chlorides	1649, 1666, 1672
Nitrates	1649, 1674
Nitrites	1674
Oil and grease	620
Phosphates	1660
Sulfates	1674
Sulfides	1639
Suspended solids	1660
Taste and odors	620, 1593, 1610, 1624, 1626, 1639, 1649, 1661
Turbidity	1593, 1624, 1649
Elemental Contaminants	
Arsenic	1062, 1652
Barium	1062
Cadmium	1652
Chromium	1062
Iron	1649
Lead	1652
Mercury	811, 1062, 1610, 1631, 1652, 1680, 1681
Selenium	1062

TABLE 104 (continued)

Contaminant	Reference Number
Silica	1604
Biocidal Contaminants	
DDT	1610, 1617, 1662, 1664, 1672
DDE	1617
Aldrin	1610, 1617, 1662
Dieldrin	620, 1610, 1617, 1662, 1664
Endrin	1610, 1617, 1662, 1664
Carbamates	1618
Chlorinated hydrocarbons	620, 1610, 1617, 1662, 1666, 1672
Organophos	1610, 1617, 1652, 1662
Herbicides	1610, 1652
Lindane	1664
Other (general)	499, 1221, 1610, 1664
Synthetic/Organic Contaminants	499, 620, 708, 1062, 1610, 1624, 1627, 1649, 1650, 1652, 1662, 1672, 1674, 1676
Biological Contaminants	
Polio Virus	1652
Virus	1652, 1657

to this plant was treated wastewater which was subsequently mixed with surface water for direct reuse. Stander and Funke (1674) reported on the effluent quality through the pilot plant. The concentration of ammonia nitrogen in the effluent was lowered from 0.3 to 0.1 mg/ℓ by passage through an activated carbon filter. ABS and BOD were also significantly reduced, 82 and 67 percent, respectively.

Table 105, from Medlar (1649), summarizes water quality analysis data from several water treatment plants in New England employing granular activated carbon filters. Two of these plants (Amesbury and Scituate) use carbon for both filtration and adsorption, while the remaining four use carbon for adsorption only. At each plant, the carbon significantly reduced ammonia levels, as between 33 and 100 percent of the influent ammonia was removed.

Phillips and Shell (1660) presented a study of the effectiveness of granular activated carbon and other general contaminants in removing BOD. The study was conducted at a pilot water plant treating wastewater effluent by chemical coagulation, filtration, and passage through 16-ft carbon columns. Data are presented in Table 106 BOD removal by the carbon columns averaged 33 percent, while COD was reduced 80 percent.

TABLE 106. ACTIVATED CARBON FILTRATION AT COLORADO SPRINGS PILOT PLANT (1660)

	Influent	Effluent	Removal
	(mg/ℓ)		(%)
BOD_5	3	2	33
COD	41	8	80
SS	4	3	25
PO_4	2	1	50
ABS	0.9	0.03	97

Activated carbon is highly efficient for removing noncolloidal, soluble, aromatic-structured color sources. David Volkert and Associates (1610) indicated that the carbon removal efficiency for color-producing substances is 100 percent of methylene-blue active substances. Table 106 shows that the color removal efficiency of carbon filters in six water treatment plants was nearly 100 percent (1649). Activated carbon has also been used industrially for decolorizing organic dye waste effluents. Recently, the nonionic polymeric adsorbents, such as Amberlite XAD-7, have been gaining popularity for this purpose (1672).

TABLE 105. SUMMARY OF WATER QUALITY ANALYSIS DATA FROM ACTIVATED GRANULAR CARBON (1649)

		Amesbury Mass.	Newburyport Mass.	Scituate Mass.	Somerset Mass.	Manchester N.H.	Burlington Mass.
COLOR:	Raw (APHA)	115	50	250	30	20	34
	Finished	0	0	5	2	0	0
TURBIDITY:	Raw (N.T.U.)	14.5	2	4	1.8	1	1.0
	Finished	0.2	0	0.12	.13	0	0.2
IRON:	Raw (mg/ℓ)	3.5	0.25	1.57	.35	0.4	0.51
	Finished	0.03	0.16	0.05	.05	trace	0.10
ODOR:	Raw	3	2	3	1	1	3
	Finished	0	0	0	0	0	0
SPECIFIC CONDUCTANCE							
(Micromhos/cm):	Raw	155	102	150	76	--	190
	Finished	230	280	268	107	--	235
Nitrate:	Raw (mg/ℓ)	0.0	0.0	0.2	0.1	--	0.4
	Finished	0.0	0.5	0.2	0.2	--	0.4
Ammonia:	Raw (mg/ℓ)	0.7	.02	0.22	.07	--	0.21
	Finished	0.0	.01	0.13	.02	--	0.14
Silica:	Raw (mg/ℓ)	11.0	4.6	7.7	.7	--	
	Finished	9.6	5.4	8.3	.8	--	

While Table 105 and the operation at Windhoek (1674) show that both organic and ammonia nitrogen were reduced by carbon adsorption, they also show that oxides of nitrogen (nitrate and nitrite) were not reduced.

Oils derived from natural, domestic, or industrial sources may occasionally be found in wastewater effluents and in water supplies. Hyndshaw (620) reported that petroleum products are very quickly adsorbed from water by activated carbon. He stated that for emergencies, such as gasoline or oil spills, large quantities of powdered activated carbon will remove the hydrocarbons.

Reduction of phosphates as PO_4 from 2 mg/ℓ in the carbon filter influent to an effluent concentration of 1 mg/ℓ is shown in Table 106 (1660). Sulfate as SO_4 was not reduced from its concentration of 220 mg/ℓ by carbon filtration at the Windhoek plant. However, sulfides in the form of hydrogen sulfide, and polysulfides that result from the reaction of chlorine with hydrogen sulfide were removed from well water, according to Lee (1639). The removal was indirectly evidenced by the destruction of sulfide odor in the water supply and resulted from a catalytic reaction with the carbon rather than from adsorption.

Activated carbon is commonly used for removing tastes and odors from water. Undesirable odors in water are caused by the vapors from many chemicals, including halogens, sulfides, ammonia, turpentine, phenols and cresols, picrates, and various hydrocarbons and unsaturated organic compounds, some of which have not been identified. Tastes and odors are also caused by substances produced by living microorganisms or decaying organic matter. Some inorganic substances, such as metal ions in high concentrations (especially iron) also impart taste and odor to water. Removal of many tastes and odors with activated carbon approaches 100 percent. Successful taste and odor removal from a municipal water supply by carbon bed filters was demonstrated in Buckingham, England (1624). The water was taken from a river that runs through farmland and had an earthy taste that resisted chlorination; clarification and sand filtration; and experimentation with ozone, permanganate, and chlorine. At Goleta, California, taste and odor in the water supply were also eliminated by passage through granular activated carbon beds (1624). Popalisky and Pogge (1661) reported that powdered activated carbon was used at a water plant treating Missouri River water to eliminate taste and odor caused by microorganic compounds. The compounds and their concentrations were not identified.

Hansen (1626) reported that installation of granular carbon filters at Mount Clemens, Michigan, completely removed the septic and phenol tastes and odors that were present in the Clinton River water supply during runoff periods. Granite City, Illinois, reported a similar experience (1593). The dosage of activated carbon required to remove taste and odor is influenced by chlorine application. Generally, carbon should be applied before chlorination (620).

Many pesticides and herbicides produce tastes and odors when present only in very small concentrations in water. Hyndshaw (620) used the beta isomer of benzene hexachloride to illustrate the influence of a small concentration of a biocide on the taste and odor quality of water. This substance, when present in amounts as little as 17 ppb in odor-free water, gives a threshold odor number (TON) of 8. (The threshold number denotes the number of volumes of odor-free water required to dilute the odor concentration to a point where the odor is just barely perceptible.) Other organic pesticides and their concentrations and subsequent effects upon taste and odor are shown in Table 107 (620).

TABLE 107. ODOR IMPARTED TO ODOR-FREE WATER BY PESTICIDES AND HERBICIDES (620)

Substance	Concentration ppm	Odor TON
Toxaphene	0.84	6
2,4 D (isoctyl)	2.0	17
2,2 D	92.5	4
D-D	0.0235	17
Rothane	50.0	Nil
Chlordane	0.07	140
BHC	0.0175	8

A simple study employing the threshold odor test gave evidence of the reduction of these compounds by activated carbon. The amount of activated carbon required to reduce the odor of each compound to a palatable level is presented in Table 108 (620).

TABLE 108. ACTIVATED CARBON REQUIRED TO REDUCE ODORS CAUSED BY PESTICIDES AND HERBICIDES TO PALATABLE LEVELS (620)

Substance	Concentration of substance ppm	Odor TON	Carbon Dosage ppm	Odor after Carbon TON
Parathion	10	50	20	4
37 gamma BHC	25	70	15	1.4
Malathion	2	50	20	4
2,4 D	6	50	40	3
DDT	5	70	4	3

Significant reduction in turbidity and suspended solids concentration is also effected with activated carbon. The data in Table 105 (1649) show that turbidity is reduced both at plants where granular activated carbon is used for filtration and adsorption and where it is used for adsorption only. Passage of water through 16-ft carbon columns effected 25 percent removal of suspended solids at Colorado Springs (Table 106, 1660).

Turbidity removal is essentially complete at Nitro, West Virginia, where activated carbon has replaced sand in the filters (1624). Final effluent JTU's are typically less than 0.1. At Granite City, Illinois, Blanck and Sulick (1593) report that suspended solids removal by carbon filtration exceeds that achieved with sand filters.

Elemental Contaminants

Although little data from municipal water purification applications are available, it appears that activated carbon can provide some removal of heavy metals. Direct adsorption provides some removal, but efficiencies can be increased to nearly 100 percent by adding an organic chelating agent (1662). The carbon removes the complex by adsorbing the organic agent, removing the metal along with it.

Patterson (1062) cited evidence that filtration of water containing 0.2 mg/ℓ arsenic through a charcoal bed yielded an effluent containing 0.06 mg/ℓ of arsenic, or 70 percent removal. He cited another report of 40 percent reduction of arsenite by activated carbon from an initial concentration of 0.5 mg/ℓ to a concentration of 0.3 mg/ℓ. Morton and Sawyer (1652) tested heavy metal adsorption during filtration through a granular bed of attapulgite clay. Data are presented in Table 109 that show the amounts of metals remaining in the effluent after filtration of various quantities of water at two different rates. The initial solutions contained 1 mg/ℓ of each metal. Arsenic was reduced from 0.97 and 0.98 mg/ℓ to 0.02 and 0.03 mg/ℓ upon filtration of 2.5 volumes of water per volume of bed.

TABLE 109. REMOVAL OF HEAVY METALS BY PERCOLATION WITH GRANULAR LOW VOLATILE MATTER ATTAPULGITE CLAY (1652)

Ratio of Volume Percolate to Volume Bed	Recoveries in Effluent			
	As	Cd	Pb	Hg
			Slow Rate*	
0	0.97 ppm	1.00	1.06	1.11
2.5	0.02	0.01	0.01	0.004
5.0	0.12 ppm	0.01	0.01	0.016
10.8	0.56 ppm	0.01	0.01	0.116
			Fast Rate†	
0	0.98	1.00	1.06	1.11
2.5	0.03	0.01	0.01	0.010
5.0	0.16	0.01	0.01	0.072
12.6	0.68	0.01	0.01	0.152

* 960 gal/ton clay/hour.
† 2,880 gal/ton clay/hour.

In a study cited by Patterson (1244), carbon adsorption did not improve barium removal efficiency over that achieved with chemical coagulation and clarification. Patterson notes that others have also found poor removal efficiency for barium.

Cadmium removal data are also presented in Table (1652). Removal of 99 percent was achieved and was unaffected by the changes in flow volume and rate.

Some success has been reported from pilot plant work on chromate removal by activated carbon. Patterson (1062) reported on a study of metal removal from municipal wastewater plant effluent, in which initial hexavalent chromium levels of 0.09 to 0.19 mg/ℓ were reduced to 0.04 mg/ℓ or less. The average effluent concentration reported was 0.017 mg/ℓ.

Initial hexavalent chromium levels of 5 mg/ℓ were reduced to 0.09 mg/ℓ following carbon adsorption. Patterson noted that it appears that activated carbon may not be equally effective at higher chromate levels.

Table 105 reports removal of iron at six water treatment plants by granular activated carbon filtration. Removal efficiencies ranged from 99 percent to 36 percent(1649).

Lead removal is shown in Table 109 for adsorption onto attapulgite clay. Over 99 percent removal was achieved at all flow volumes and rates tested.

Mercury adsorption has been studied extensively. In Table 109 , it is shown that adsorption onto attapulgite clay of over 99 percent was achieved after filtration of 2.5 volumes of water per volume of bed. Removals decreased to 86 percent removal as 12.6 volumes per volume of bed were filtered. Logsdon and Symons (811) conducted jar tests using powdered activated carbon. Increasing carbon dosages increased removal of both inorganic and organic (methyl) mercury. Organic mercury removal was more efficient than inorganic mercury removal for a given dosage of powdered activated carbon at effluent concentrations less than 2 ppb. Residual concentrations of 0.8 ppb of inorganic and 0.2 ppb of methyl mercury were achieved. Further tests were performed in which powdered carbon was added just before alum coagulation to improve mercury removal. Removal by alum alone was about 40 percent, whereas removal with 65 mg/ℓ of activated carbon plus 30 mg/ℓ of alum was in excess of 70 percent with an initial mercury concentration of 9.3 ppb.

The capacity of granular activated carbon for removing mercury from water was also evaluated by pumping mercury solutions through columns (811). Average influent concentrations ranged from 20 to 29 ppb. Mercury removals declined as the number of bed volumes (based on gross void space) of water treated increased. Columns with 3.5-min contact times removed 80 percent of the inorganic mercury for up to 15,000 bed volumes of water and 80 percent of the methyl mercury for up to 25,000 bed volumes treated. Evaluation of carbon column performance at 80 percent removal indicated that as contact time in the column increased, the amount of mercury adsorbed by the carbon increased. Also, granular activated carbon adsorbed more methyl mercury than it did inorganic mercury per gram of adsorbent. This behavior was expected, since activated carbon has a high capacity to adsorb organics.

Thiem (1680)also conducted jar tests on mercury adsorption using powdered activated carbon. Solutions containing 10 ppb mercury were brought into equilibrium with various carbon dosages. Less than 30 percent of the mercury was adsorbed at a carbon dosage of 10 mg/ℓ at pH 7. When 100 mg/ℓ of carbon was applied to the test solution, removal approached 80 percent. Removal decreased with increasing pH, with best removal occurring at pH 7. The addition of chelating agents such as EDTA or tannic acid prior to contact with the carbon increased adsorption. Concentrations of as little as 0.02 mg/ℓ of EDTA or 1 mg/ℓ of tannic acid increased removals from 10 to 30 percent, depending upon the carbon dosage that was applied and the pH of the system.

Patterson (1062) summarized the data on activated carbon removal of mercury, saying that the highest percentage removals (80 to 95 percent) are achieved with more concentrated mercury solutions. However, lowest effluent mercury results from treatment of less concentrated waters, although the relative efficiency is less. Thus carbon treatment of initial mercury below 1 ppb yields an efficiency of removal of less than 70 percent, but effluent mercury below 0.25 ppb. Carbon treatment of initial mercury concentrations of 5 to 10 ppb yields about 80 percent removal and effluent levels of below 2 ppb. Carbon treatment of initial mercury concentrations between 10 and 100 ppb yields 90 percent or greater efficiency.

Poor selenium removal from well water by activated carbon has been reported in one case cited by Patterson (1062). Removal was less than 4 percent. Poor or no reduction of silica has also been shown by the data from six operating water purification plants employing carbon adsorption and filtration(1649).

Biocidal Contaminants

As with other synthetic-organic compounds, some of the organic pesticides and herbicides that are resistant to removal by conventional treatment techniques are effectively removed by adsorption. David Volkert and Associates (1610). cited evidence that over 99 percent of the following chlorinated hydrocarbons can be adsorbed by activated carbon:

- DDT
- Aldrin
- Dieldrin
- Endrin
- Chlordane
- Heptaclor epoxide

- Lindane

- Methoxychlor

- Toxaphene

Laboratory studies cited by Stone (1662) have shown that the following reductions in chlorinated hydrocarbon concentrations can be achieved by contacting with appropriate doses of activated carbon (Table 110):

TABLE 110. ACTIVATED CARBON REMOVALS OF CHLORINATED HYDROCARBONS ACHIEVED IN LABORATORY EXPERIMENTS (1662)

Substance	Initial Concentration	Treated Concentration	Percent Reduction
Chlorinated hydrocarbons	6.3 ppb	0.04-0.11 ppb	98-99
DDT	5 ppm	-	90
DDT	5 ppm	-	76
Aldrin	6.6 ppb	-	90
Aldrin	5 ppm	-	85
Aldrin	-	-	99
Dieldrin	-	-	99
Dieldrin	0.5-10 ppb	0.25 ppb	50-97
Dieldrin	4.4 ppb	-	99
Endrin	5 ppm	-	86
Endrin	0.5-10 ppb	0.25 ppb	50-97
Chlordane	50 ppm	-	99
Lindane	10 ppb	1 ppb	90
Lindane	25 ppb	-	90
Lindane	1 ppb	0.05 ppb	95

Activated carbon removals of several pesticides are well illustrated by results of laboratory studies cited by the California State Water Resources Control Board (1323) shown in Table 111.

TABLE 111. REMOVAL OF SPECIFIC TOXIC MATERIALS BY CARBON ADSORPTION (1323)

Carbon Dosage (mg/ℓ)	Residual (ppb) Aldrin	Endrin	Dieldrin	DDT	DDD	DDE	Toxaphene
Control	48	62	19	41	56	38	155
1.0	--	--	--	41	--	34	147
2.0	26	15	6.3	--	6.9	--	80
2.5	--	--	--	21	--	29	--
5.0	15	3.4	2.4	3.7	3.7	12	31

TABLE 111. (continued)

Carbon Dosage (mg/ℓ)	Residual (ppb) Aldrin	Endrin	Dieldrin	DDT	DDD	DDE	Toxaphene
10.0	12	1.5	1.1	--	2.2	--	2.7
12.5	--	--	--	<1	--	3.3	--
25.0	6.3	0.56	--	--	0.45	1.1	--
50.0	4.4	0.22	--	--	0.35	0.9	--

Adsorption of the carbamate pesticides Sevin and Baygon on granular activated carbon was investigated by El-Dib et al. (1618). Passage of a 5 mg/ℓ solution of Baygon through carbon columns effected complete removal for up to 273 bed volumes of water when a contact time of 3.76 min was allowed. In the case of Sevin, 1,800 bed volumes were passed with complete removal under the same conditions. Rapid breakthrough of the pesticides into the effluent occurred, however, as the contact times were decreased to 1 or 0.5 min.

According to reference 198 the available data on organophosphorus pesticide removal indicates that the efficiency of activated carbon ranges from 50 to over 99 percent. Stone and Company (1662)cited laboratory-scale tests showing that Parathion was reduced from 10 to 2.5 mg/ℓ using 20 mg/ℓ powdered carbon; and from 11.4 to 0.05 mg/ℓ using dual granulated carbon filters. Malathion was reduced in laboratory-scale tests from 2 to 0.25 mg/ℓ with 10 mg/ℓ powdered carbon. Some data on removal efficiencies of the organic herbicides were also cited. They report that over 99 percent removal of 2,4-5-TP and 2,4-D is possible.

Robeck et al. (1168) surveyed the effectiveness of various water treatment processes in pesticide removal. Table 112 summarizes their results using carbon in both a slurry form and in beds.

TABLE 112. SUMMARY OF CUMULATIVE PESTICIDE REMOVAL AT 10-ppb LOAD

Process	Pesticide Removed - Percent DDT	Lindane	Parathion	Dieldrin	2,4,5-T Ester	Endrin
Carbon:						
Slurry						
5 ppm		30	> 99	75	80	84
10 ppm		55	> 99	85	90	90
20 ppm		80	> 99	92	95	94
Bed 0.5 gpm/cu ft	> 99	> 99	> 99	> 99	> 99	>99

In addition to activated carbon, other adsorbents such as clays and synthetic polymeric adsorbents are capable of removing biocidal contaminants. Morton and Sawyer(1652) studied the adsorption of diazinon, an organophosphorus pesticide, onto attapulgite clay. Coarse-ground, high-volatile matter attapulgite was stirred with contaminated water in laboratory experiments. At least 50 percent of the diazinon in a 0.1 mg/ℓ solution was removed by the clay in a 10 percent clay suspension. An investigation of the use of the synthetic polymer Amberlite XAD-4 for the removal of various pesticides was cited by Stone and Company (1662) The pesticides examined included -BHC, lindane, B-BHC, aldrin, and dieldrin in tap water at initial concentrations of 1 ppb each. Consistent removals of over 60 percent were reported.

Schwartz (1221)also studied the adsorption of selected pesticides on activated carbon and mineral surfaces. He found that the clay minerals ilite, kaolinite, and montmorillonite suspended in dilute pesticide solutions adsorbed very little 2,4-dichlorphenoxyacetic acid (2,4-D) or isopropyl N-(3-chlorophenyl) carbamate (CIPC). Adsorption of CIPC from aqueous solution with powdered activated carbon, however, was extensive (90+ percent).

Synthetic/Organic Contaminants

Adsorption is commonly cited as a presently available technology for removing particulate, colloidal, and soluble organic contaminants from water. Many of the organics present in water supplies - particularly the soluble and colloidal organics - are of a refractory nature, i.e., they resist removal by conventional methods. A number of these are potentially toxic or carcinogenic and, as a result, their detection, identification, and treatment in water is receiving increasing attention. These substances, even in small amounts, contribute to taste and odor conditions and may pose a chronic health hazard. As has been discussed, activated carbon is widely applied for taste and odor removal; however, its effectiveness for removing residual organics has just come under study in recent years. The delay has been caused in part by the lack of standard procedures for identifying and classifying the vast assortment of organics that occurs in trace quantities in water (499, 1650). The delay is also due to the search for a gross organic parameter that can be used as a measure of organics. The more common parameters used include carbon chloroform extract, liquid extraction, paper and gas chromatography, fluorescent spectroscopy, and radiation.

Traditionally, carbon life expectancy has been based on the capability of the carbon to absorb tastes and odors. But research has shown that the life expectancy of carbon to reduce carbon chloroform extract or organic compounds is somewhat less than that to remove tastes and odors (1649). Medlar (1649) suggested that monitoring carbon chloroform extract (CCE) concentration in carbon filtered water would provide a conservative estimate of filter performance, but noted that the CCE test may not encompass all the compounds that should be considered.

Carbon chloroform extract indicates the presence of stable organic compounds in water. The extract has an operational definition and is a mixture of organic compounds that can be adsorbed onto activated carbon and then desorbed with organic solvents under specific controls. Examples of substances measured with this method include substituted benzene compounds, kerosene, polycyclic hydrocarbons, phenylether, and insecticides. The efficiency of activated carbon in reducing CCE depends upon several factors including water temperature, initial amount of contaminant, and the molecular weight of the contaminant (1610). Percentage removals must therefore be determined by laboratory testing. Removals ranging from 50 to 99 percent were reported by David Volkert and Associates (1610).

The adsorption of polycyclic (polynuclear) aromatic hydrocarbons (PAH) from water by activated carbon was discussed by Harrison et al. (1677). These compounds are potential carcinogens under certain conditions. Carbon adsorption has been shown to give 99 percent removal of PAH from water filtered by prior seepage through river bank soil.

Bis-ethers are synthetic organic compounds that may occur in water associated with industrial discharges. Stone and Company (1662) cited laboratory tests of activated carbon treatment in which isopropyl ether concentrations were reduced from 1,023 to 20 mg/ℓ, butyl ether concentrations from 197 mg/ℓ to nil, and dichloroisopropyl ether concentrations from 1,008 mg/ℓ to nil.

The treatment of dilute phenolic industrial wastewater was reviewed by Patterson (1062). Adsorption onto activated carbon has been employed to remove over 99 percent of the phenol present in process waters with initial concentrations ranging from 5,325 mg/ℓ to 0.12 mg/ℓ. Final phenol concentrations ranged from 0.25 mg/ℓ for treatment of a concentrated solution to 0.001 mg/ℓ for treatment of weaker solutions.

Foaming agents such as linear alkyl benzene sulfonate in concentrations up to 5 mg/ℓ can be removed by activated carbon with 90 to 100 percent efficiency according to evidence cited by David Volkert and Associates (1610). Table 106 (1660), which presents data on activated carbon filtration at the Colorado Springs pilot plant, shows 97 percent removal of alkyl benzene sulfonate (ABS). Stander and Funke (1674) reported reduction of ABS from 4 to 0.7 mg/ℓ at the Windhoek pilot plant. Organic acids are also reported to have been reduced from 1 to 0.4 mg/ℓ.

Morton and Sawyer (1652) studied the adsorption of two organic compounds - diethylstilbestrol (DES), which is a hormone fungi and aflatoxin, which is a natural toxin produced by fungi onto attapulgite clay. Attapulgite is a magnesium aluminum silcate clay that exhibits a high debree of adsorption for low-weight organic molecules. Coarse-ground, high-volatile-matter attapulgite was contacted with contaminated water in laboratory experiments. DES at a concentration of 5 ppb was decreased 68 and 76 percent by contacting with 1.1 and 10 percent (by weight) clay suspensions, while in a 50 ppb solution the removals were 68 and 89 percent, respectively. More than 98 percent of the aflatoxin at concentrations of 0.5 ppb and 5.0 ppb was removed by both 1.1 and 10 percent clay suspensions. The results of column percolation studies through granular low-volatile-matter clay are presented in Table 113.

TABLE 113. REMOVAL OF ORGANICS BY PERCOLATION WITH GRANULAR, LVM ATTAPULGITE

Volume rates percolate to bed volume	Recoveries in Effluent	
	Slow rate*	Fast rate†
	50 ppb DES Solution	
0	52 ppb	48 ppb
2.5	ND‡	ND
5.0	ND	1
10.8	ND	--
12.6	--	2
	20 ppb Aflatoxin Solution	
0	17 ppb	17 ppb
2.5	ND	ND
5.0	ND	ND
10.8	ND	--
12.6	--	ND

* 960 gal/ton clay/hour.
† 2,880 gal/ton clay/hour.
‡ ND = not detectable.

Stone and Company (1662) cited data concerning the treatment of synthetic-organic polychlorinated biphenyls (PCB's) by adsorption onto clay minerals and Amberlite polymeric adsorbents. In laboratory tests Amberlite XAD-4 removed up to 76 percent of the PCB present in solutions. Several clay minerals demonstrated PCB removal capability in laboratory tests: illite - 60 percent, montmorillonite - 40 percent, and kaolinite - 40 percent. Kinoshita and Sunata (708) evaluated the adsorption of PCB onto powdered activated carbon in a jar test and found that the initial concentration of 100 ppb PCB was reduced to 10 ppb in the product water.

The Amberlite adsorbents represent a new technology for adsorbing organic molecules from water. They are used specifically for adsorbing aromatic and aliphatic compounds. According to Simpson (1672) small molecules such as phenol are effectively adsorbed by Amberlite XAD-4, while for a larger molecule such as an alkylbenzene sulfonate, Amberlite XAD-2 has a much higher adsorptive capacity. Phenol is a model aromatic, low-molecular weight compound that is considered to be highly objectionable, as are some of the chlorinated phenol products. Using the XAD-4 adsorbent, up to 40 bed volumes of a 500 ppm water solution of phenol were treated with less than 10 percent leakage at a flow rate of 0.5 gal/ft^3/min. At a higher flow rate of 2.0 gal/ft^3/min, 20 bed volumes were treated with less than 10 percent leakage. It was further found that the adsorptive capacity was higher for chlorinated phenols than for simple phenol. Simpson (1672) cited a laboratory study in which the removal efficiency of Amberlite XAD-2 for a list of organics at flow rates of 1.25 gpm/cu ft were determined. The results are presented in Table 114. Nonionic compounds were removed with 100 percent efficiency while ionized compounds were less effectively removed.

In the nationwide study of water supplies and water treatment facilities by Symons et al. (1676), it was concluded that both powdered and granular activated carbon treatment significantly reduced the trace concentrations of total trihalomethane in the product water.

TABLE 114 . ADSORPTION OF ORGANIC COMPOUNDS ONTO AMBERLITE XAD-2 POLYMERIC ADSORBENT (1672)

1. Aliphatics	Influent	Effluent	Retention Efficiency, %
a) alcohol: n hexanol	200 ppm	30 ppm	85
b) ester: ethyl butyrate	100	0	100
c) ketone: methylisobutylketone	100	0	100
2. Aromatics			
Benzene	100	0	100
Benzene sulfonic acid	3.0	2.1	31

TABLE 114. (continued)

2. Aromatics	Influent	Effluent	Retention Efficiency, %
p-toluene sulfonic acid	9.0	6.9	23
Benzoic acid	1.0	0.8	23
Benzoic acid (pH 3.2)	1.0	0	100
Phenol	0.4	0.22	45
Phenol (Amberlite XAD-7)	0.4	0.06	86
O-Cresol	0.3	0	100
2, 4-dimethyl phenol	0.4	0	100
p-nitrophenol	0.2	0	100
2-methylphenol	0.3	0	100
4, 6-dinitro-2-aminophenol	0.4	0.22	43
Phenylenediamine	0.9	0.02	98
Aniline (Amberlite XAD-7)	4.0	0	100
Naphthalene	0.05	0	100
2-hydroxy-3 naphthoic acid	0.6	0.37	39

Biological Contaminants

The limited efficiency of activated carbon in removing viruses from wastewater was discussed in the advanced wastewater treatment section of this report. Results show that activated carbon is inefficient in removing viruses from drinking water. Oza and Chaudhuri (1657) suggested that the inefficiency may be due to the exclusion of viruses from the micropores of activated carbon because of their size. Coal adsorption of a bacterial virus was investigated, and results indicated that coal may be a more effective adsorbent than activated carbon. Morton and Sawyer (1652) demonstrated that attapulgite clay also has the capacity to adsorb polio virus from water. Aqueous virus-clay suspensions were shaken for 1 min and then filtered. High-volatile-matter clay completely removed virus infectivity from a 20 percent clay suspension with an initial virus concentration of 16 million infectious particles per mℓ. However, reducing the contact time from 30 to 5 min or the clay concentration from 20 to 5 percent resulted in incomplete removal.

ION EXCHANGE

Introduction

Ion exchange has its greatest current application in industrial and small-scale potable water supply operations. The most common use of ion exchange is for removal of hardness (calcium and magnesium cations) from municipal, industrial, household, and laboratory water supplies. It is particularly suited for desalting brackish water, pretreating water that must be almost completely demineralized for industrial use, and removing metals from industrial metal plating rinse wastewaters.

No one ion exchange resin is capable of removing all ionic contaminants. Various resins, depending upon their chemical nature, show preferential selectivity for specific ions. Table 115 presents data on ion selectivity for various types of exchange resins (1062).

TABLE 115. ION EXCHANGE RESINS SELECTIVITY (1062)

Resin	Resin Selectivity*
Strong-acid cation (Sulfonic)	Li^+, H^+, Na^+, NH_4^+, K^+, Rb^+, Cs^+, Mg^{+2}, Zn^{+2}, Cu^{+2}, Ca^{+2}, Pb^{+2}
Weak-acid cation (Carboxylic)	Na^+, K^+, Mg^{+2}, Ca^{+2}, Cu^{+2}, H^+
Strong-base anion (Type I)	F^-, OH^-, $H_2PO_4^-$, HCO_3^-, Cl^-, NO_2^-, HSO_3^-, CN^-, Br^-, NO_3^-, HSO_4^-, I^-
Strong-base anion (Type II)	F^-, $H_2PO_4^-$, OH^-, HCO_3^-, Cl^-, NO_2^-, HSO_3^-, CN^-, Br^-, NO_3^-, HSO_4^-, I^-
Weak-base anion	F^-, Cl^-, Br^-, I^-, PO_4^{-3}, NO_3^-, CrO_4^{-2}, SO_4^{-2}, OH^-

*Increasing selectivity left to right.

The more strongly a resin adsorbs a particular ion, the more complete the ion removal. However, a high affinity between a particular resin and a specific ion also means greater difficulty in regenerating the resin; that is, it is more difficult to release the adsorbed ions to make the resin reuseable. As a result, regeneration of an effective resin is seldom carried to completion, and the operational capacity of an ion exchange resin may be reduced to 50 to 60 percent of theoretical capacity.

Ion exchange processes are very sensitive to both clogging and fouling. An ion exchange resin bed is a good filter; therefore, suspended solids in water will clog the bed. Fouling results when the resin adsorbs materials which, because of their adsorption or absorption into the resin pores, cannot be removed in the regeneration step. Fouling often results from the irreversible sorption of high molecular weight organic acids.

The nonselective nature of most exchange resins is a drawback when attempting to remove a low-level contaminant from water. The simultaneous removal of other nontarget ions rapidly increases the cost of using the process. More selective resins are currently becoming available (1634).

In the past, application of ion exchange processes was confined to the removal of ionic contaminants. Recently, however, some ion exchangers have been developed that can remove nonionic species, e.g., Al_2O_3, Fe_2O_3, H_2SiO_3, etc. Resinous adsorbents are also available that are particularly suited for removing organic compounds - including biocidal and synthetic/organic contaminants - from water. While often used in conjunction with true ion exchange processes, the mechanism of organic removal is actually adsorption; therefore, the resinous or polymeric adsorbents (as they are called) are discussed in the carbon adsorption section of this report. The literature utilized during this review on the removal of various contaminants by ion exchange methods is indicated in Table 116.

Water Quality Parameters

Reduction of total dissolved solids may be an important application of the ion exchange process when reclaimed wastewater for potable reuse is the objective of treatment. High total dissolved solids concentrations, from 500 to over 1,000 mg/ℓ, are found in wastewaters. Approximately 300 mg/ℓ of total dissolved solids is generally considered the increment added during one cycle of domestic use of a water supply. Ion exchange treatment of water and wastewater for removal of dissolved solids is most successful after prior treatment using both conventional and advanced techniques has taken place. This prevents clogging and fouling of the exchange resins.

Ion exchange is technically capable of producing a water with only 0.055 micromhos (μmhos) of specific conductance. One micromho will normally indicate a dissolved solids concentration of 0.5 to 0.6 mg/ℓ. However, water of such purity is rarely

TABLE 116. LITERATURE REVIEWED PERTAINING TO ION EXCHANGE

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	1062, 1598, 1659, 1661
Chlorides	1062, 1610, 1659, 1661
Color	1610
Cyanides	1062, 1610
Fluorides	1062, 1610
Hardness	1659
Nitrates	165, 1592, 1598, 1602, 1610, 1622, 1630
Phosphates	1592, 1602
Sodium	1602, 1610, 1659
Sulfates	1602, 1610, 1659
Total dissolved solids	1062, 1592, 1602, 1610
Elemental Contaminants	
Arsenic	1062, 1244, 1596
Barium	1062
Boron	1062
Cadmium	1062
Chromium	1062
Copper	1062, 1634
Iron	1062, 1589
Manganese	1062, 1589
Mercury	1062, 1634
Nickel	1062

TABLE 116 (continued)

Contaminant	Reference Number
Selenium	1062
Zinc	1062, 1634

required, and the costs to achieve such purity would be prohibitive. Complete demineralization of municipal water supplies is unwarranted and may even have adverse consequences, as the body requires certain trace concentrations of many minerals.

The EPA and the LA County Sanitation Districts jointly funded an advanced wastewater treatment facility in Pomona, California. The results of the ongoing tests have been comprehensively reported by Parkhurst (1659) and Chen (1602). The ion exchange system, which was proven to be of economic and technical feasibility, effectively reduced the total dissolved solids by 90 percent. The treatment involved passage first through a primary cation column and an anion column, then through a secondary cation column and a secondary anion column. A series of 80 complete operating cycles including exchange, backwashing, and regeneration was completed at a 2.5-gpm flow rate. The total dissolved solids concentrations of the feed and product waters averaged 610 mg/ℓ and 72 mg/ℓ, indicating an 89 percent average reduction. A complete listing of the various constituents at successive stages of the demineralization sequence is presented in Table 117 (1659).

All of the major anions and cations were removed to a considerable degree except silica. Silica removal can be accomplished, however, with highly basic anion resins (1592). Calcium and magnesium, the cations responsible for water hardness, are almost completely removed in the primary cation exchanger; but only about half of the sodium, potassium, and ammonium ions are removed in this column. The secondary cation exchanger, however, efficiently removed the majority of the remaining ions. Similarly, the primary anion exchanger removed most of the sulfate ions, while the nitrate, chloride, and orthophosphate ions were partially removed by both the primary and secondary stages.

David Volkert and Associates (1610) cited evidence that the sodium and fluoride concentrations of water can be reduced by 95 percent using ion exchange. It reported that chloride, sulfate, and nitrate can be reduced by as much as 95 percent, depending on the degree to which the exchange resin can be regenerated.

TABLE 117. AVERAGE WATER QUALITY CHARACTERISTICS OF THE ION EXCHANGE PILOT PLANT UNDER TYPICAL OPERATING CONDITIONS (1659)*

	Carbon Column Effluent (feed)	Primary Cation Effluent	Primary Anion Effluent	Secondary Cation Effluent	Secondary Anion Effluent (product)
Calcium	53†	2.0	1.7	1.1	0.60
Magnesium	17	0.59	0.56	0.38	0.00
Sodium	126	61	59	16	15
Potassium	14	7.3	7.1	1.9	1.9
Ammonia, as N	20	9.6	9.2	4.0	3.8
Sulfate	72	72	3.6	3.6	1.3
Nitrate, as N	2.9	2.8	1.6	1.5	0.35
Chloride	135	132	84	83	14
Orthophosphate, as PO_4	27	27	15	14	0.25
Total alkalinity, as $CaCO_3$			51		39
pH	7.4	2.7	5.7	2.8	5.8
Conductivity	10	9.7	6.8	5.5	3.7
Silica, as SiO_2	23				23

*Data taken from May 1968 through December 1968.
†All constituents in mg/ℓ except: (1) pH and (2) conductivity (μmhos/cm).

The application of ion exchange to nitrate reduction of reclaimed wastewaters may be particularly important, as these waters frequently may contain nitrate concentrations in excess of the 10 mg/ℓ limit set by the U.S. Environmental Protection Agency as an interim primary drinking water standard. The use of ion exchange resins for the removal of ammonium and nitrate ions has been discussed in the adsorption section of this report. Few specific exchange resins are available for removal of the nitrate ion from wastewater; ammonium ion removal with certain specific ziolite resins can be quite effective. In wastewater treatment, significant nitrogen removal may be effected by removing ammonia with ion exchange; in water purification treatment, nitrate is often the most significant form of nitrogen. Strong nonspecific anion exchange resins have been applied to the removal of nitrates from drinking water; but because the resins are nonspecific, competition from sulfides, chlorides, silicates, and phosphates limits the target removal of nitrates.

This problem of competing ions may be significant when considering potable reuse, since reclaimed wastewaters that are high in nitrates are also likely to be high in these anions. The caking and clogging problems caused by iron, turbidity, and colloidal matter are also significant. In other words, ion exchange technology for nitrate removal is still developing. A successful application of the available technology has taken place at Long Island, New York (1622, 1630). A few years ago, Long Island communities began experiencing rising nitrate levels in their drinking water supply source. Drinking water was obtained from groundwater wells that had been contaminated with nitrates from septic tanks and cesspools. An ion exchange process originally developed to demineralize industrial process water was adapted in a prototype ion exchange plant; it reduced the nitrate content of several thousand gallons of the Long Island water from 22 ppm to 0.5 ppm nitrate. A plant was then built that successfully provided ion exchange treatment for the Long Island well water under the constant flow and pressure requirements of a municipal water system. However, the well water was of relatively low total dissolved solids, and it is uncertain how this process would perform on waters of poorer quality.

A water quality parameter that is of interest in potable water treatment is color. According to David Volkert and Associates (1610), ion exchange resins have been developed that will completely remove color-causing organic dye wastes, humates, and lignates.

Elemental Contaminants

Ion exchange techniques have been applied to the treatment of industrial process water containing trace metals for several years. Application of these techniques to drinking waters containing trace concentrations of these metals is analogous. In a review of the literature, David Volkert and Associates (1610) found that several trace elements can be removed from water to a level of 95 percent. Table 118 lists these elements and gives the maximum concentrations that can be reduced to 1974 Standards and Guidelines in a single pass through an ion exchange process (1610). Water containing concentrations of a particular contaminant higher than the maximum listed in the table can be either passed through a series of ion exchange columns with different types of resins, or pretreated by a method such as lime coagulation to precipitate the major amount of the metal present.

Calmon (1596) noted that anion exchange treatment can be used to remove residual arsenic after lime coagulation is used to precipitate the major amount present. Both weak and strong-base ion exchange resins appear effective in removing arsenate and arsenite from drinking water (1062). Calmon (1596), treating an arsenate water containing 68 mg/ℓ arsenic at pH 6.95 with a weak-base anion exchange resin (Ionac A-260), reported 82 to 100 percent removal. Medium and strong-base resins (Ionac A-300, A-540, and A-550) were less effective.

TABLE 118. REMOVAL OF TRACE ELEMENTAL CONTAMINANTS FROM WATER BY ION EXCHANGE (1610)

Treatment	Contaminant	% Removal	Maximum* Concentration
Ion Exchange	Arsenic	95	2.0 mg/ℓ
	Barium	95	20.0
	Cadmium	95	0.2
	Chromium	95	1.0 mg/ℓ
	Copper	95	20.0
	Cyanides	95	4.0
	Lead	95	0.1
	Iron	95	6.0
	Manganese	95	0.1
	Mercury	95	0.04
	Selenium	95	0.2
	Silver	95	0.1
	Zinc	95	100

*Maximum concentration in raw water, which can be reduced to 1974 Standards and Guidelines in a single pass through process. If raw water concentrations are higher, then combination or duplication of processes or other processes must be considered.

Again using the weak-base anion exchange resin (Ionac A-260), Shen (1244) treated synthetic water containing 106 mg/ℓ of arsenic. Only 20.7 percent of the arsenic was removed. When well waters with naturally high arsenic levels were treated, essentially 100 percent removal was achieved. The disparity of the results was not explained.

Bone char and activated alumina readily remove arsenic via an ion exchange mechanism. Arsenic sorption on bone char results in an irreversible change in the chemical structure of the char. Consequently, exhausted bone char must be discarded; it cannot be regenerated. Activated alumina is regenerable.

Effective removal of barium by ion exchange has been reported. Patterson (1062) cited a 98.5 percent reduction of barium from 11.7 to 0.17 mg/ℓ in a full-scale ion exchange groundwater softening plant in which a general nonspecific water softening resin was used.

According to Patterson (1062), ion exchange is a common method for recovery of cadmium from industrial wastewaters, and many exchange resins are available with high specificity for the metal. To meet stringent effluent standards, some industries are using ion exchange rather than cheaper but less effective methods to treat chromate and chromic acid waters. With proper pH adjustment, chromate is removed even in the presence of high concentrations of sulfate and chloride (1062). Reduction of hexavalent chromium to 0.023 mg/ℓ in a metal finishing wastewater has been reported (1062). Cation exchange can be applied to remove trivalent chromium, and anion exchange can be employed to remove chromate and dichromate (1062).

Ion exchange is capable of achieving very high levels of copper removal. Reduction of 1.02 mg/ℓ copper to less than 0.03 mg/ℓ has been reported. A selective ion exchange resin (Amberlite IR 120) reduced the copper concentration of an industrial copper plating rinse solution from 45 mg/ℓ to an undetectable amount (1062). Koerts (1634) found that ion exchange can remove copper and zinc from industrial waters to produce effluents containing as little as 0.04 mg/ℓ of copper and 0.1 mg/ℓ of zinc.

Removal of several metals, such as iron, manganese, lead, copper, and nickel can be accomplished with ion exchange, but the processes involve low pH or anaerobic water streams, which make them normally unsuitable for municipal water treatment. This technology, however, may be expected to develop as more attention is given to the application of ion exchange for the purification of reclaimed municipal wastewaters.

Ion exchange treatment of inorganic mercury-bearing waters appears to be capable of furnishing an effluent with 1 to 5 ppb inorganic mercury. Table 119 from Patterson (1062) reviews the experience with ion exchange treatment for inorganic mercury. Effluent values in the ppb range are indicated. Preliminary tests have indicated that cation and anion exchange resins in series can remove 98 percent of both inorganic and organic mercury forms. Akzo Chemie has developed a special resin for mercury, which produces effluent levels below 5 ppb mercury (1634).

TABLE 119. ION EXCHANGE TREATMENT FOR INORGANIC MERCURY (1062)

Resin Type	Treatment pH	Mercury (µg/ℓ) Initial	Mercury (µg/ℓ) Final	Additional Treatment	Percent Removal
Mtylon-T	5-6	5,000-25,000	1	--	100
Anion*	7	850	2.5	--	99.7
Macroreticular	na	10,000	<10	--	100
Anion*	na	470	30	--	94
Osaka IE*	"acidic"	3,000-10,000	100-150	Prefilter	98
Osaka MR*	na	100-150	2-5	Osaka IE Resin	98
Activex	na	10	<5	--	50
Ajinomoto	1.5	60	5	--	92
	6.0	87	3	--	97
	11.0	1,800	990	--	45
Billingsfors	6.5	35	1	--	97

*Mercury removed as mercuric chloride complex, $HgCl_x^{(x-2)}$.

Patterson (1062) presented evidence that cation plus anion exchange applied to secondary wastewater effluent can remove selenium to a level of 99.7 percent. He also cited an example of 85 percent removal of silver from an extremely dilute secondary sewage effluent by cation exchange, and 91.7 percent removal by combined cation-anion exchange.

Calmon (1598) discussed the use of ion-exchange for trace heavy metal removal. As part of his paper, he listed the specific resins that can be used to remove various heavy metals. These are as follows:

Element	Polar Group
Arsenic (3+)	Fluorone
Beryllium	Phosphonic Diallyl phosphate
Bismuth	Pyrogallol
Boron	N-methyl glucamine, tris hydroxymethyl amino methane
Cesium	Phenolic OH + Sulfonic groups
Cobalt	M-phenylene diamine, 8-hydroxyquinoline
Copper	Phenolic OH + phosphonic groups, 8-hydroxy quinoline m-phenylene diamine imino diacetic acid alginic acid
Germanium	Phorone
Gold	Pyridinium, thiourea

Element	Polar Group
Iron	Alginic acid m-phenylene diamine hydroxamic acid phosphonous phorone chlorophyll haemin deriv.
Lead	Pyragallol, Phosphoric
Mercury	Thiourea, thiol, iminodiacetic acid, mercapto resins
Nickel	Alginic acid, Dimethylglyoxime
Potassium	Dipicrylamine
Strontium	Phosphorous
Titanium	Chromotropic acid
Uranium	Pyridinium, phosphorous ester
Viruses	Metal cation proteins
Zinc	Anthranilic
Zirconium	Phosphate ester

He also discussed the various types of exchangers, dosages, operating problems, etc., that can be anticipated for removal of each metal.

Biocidal Contaminants

Ion exchange units are not specifically designed to remove biocidal contaminants. However, some incidental removal of these pesticides, insecticides, and herbicides may occur due to adsorption on the resin material. This phenomenon will occur until the adsorptive capacity of the resin is exhausted.

Synthetic/Organic Contaminants

Because many of the synthetic/organic contaminants have chemical and physical properties that are similar to the biocidal contaminants, the same considerations apply here as in the biocidal section.

Biological Contaminants

Since the primary removal mechanism of ion exchange is based on particle charge properties, any removal of the biological contaminants will be incidental. Any removal that does occur will probably be due to mechanical filtration.

REVERSE OSMOSIS

Introduction

For readers unfamiliar with the osmotic membrane process, one can picture these membranes as thin sheet filter materials. Initial investigators thought that the membranes acted as strainer-type materials with such small pore sizes that most ionic and biological molecules were too large to pass through. Recent studies with electron microscopes have shown that removals are controlled by molecular diffusion through the membrane and that salts and other impurities diffuse much more slowly than water (1682). The differential pressure to provide a driving force through the membrane is supplied by pumps. The wastewater bleed of concentrated contaminants is continuous, and its volume normally equals from 5 to 30 percent of the volume of the process influent volume. Membrane fouling by suspended solids, organic slimes, and precipitates is a problem unless substantial pretreatment of the influent water is provided. Therefore, reverse osmosis is normally located as the last unit process in the water treatment chain.

Very little actual operating data are available regarding the use of reverse osmosis (RO) for treatment of municipal water supplies, because the quality provided by reverse osmosis systems has not as yet been required. Even in areas of high TDS that could benefit from RO treatment, the cost of this unit process has been prohibitive. There has, however, been a substantial amount of research conducted on the use of RO for polishing highly treated wastewaters. This research, though, primarily focuses on the removal of selected contaminants from solution in laboratory and pilot scale projects. Since these are similar in many respects to raw water supplies, performance is often analogous to water treatment situations.

Reverse osmosis may become more important in the future as both a tertiary wastewater and raw water supply treatment process, because of its capability to remove a high percentage of all types of general, elemental, and biological contaminants as well as many synthetic/organic and biocidal constituents. With the recent concern over even very small concentrations of heavy metals, residual organics, and toxic compounds in water supplies, RO with its 99$^+$ percent removal efficiency may see increased usage, albeit at a high cost.

Aqueous solutions of two or more solutes reacting with a membrane may give product flux and solute retentions that are quite different from those predicted from the behavior of solutions of the individual organics. This results from interactions between solutes, their products, and the membrane. One interesting result in this regard is that improved retentions of solutes were observed when mixed in partially treated wastewaters, thus increasing the attractiveness of wastewater RO treatment for reuse (367). Improved retentions are thought to be due to synergistic effects among the high molecular weighted components.

Various types of membrane systems have been tested on a variety of wastewater effluents from primary to highly treated tertiary. Laboratory tests have also been conducted on the removal of many synthetic/organic chemicals and biocidal compounds. The RO process does not destroy any of the input contaminants, but only separates them into two streams, with the waste stream containing the rejected materials. Depending upon the feed conditions and the desired objectives, the product water volume can range up to 95 percent of the total influent, with the remainder representing the discarded flow. The characteristics of RO membranes can be controlled within a wide range by controlling the manufacturing variables. In general, as one improves the contaminant removal efficiencies, the flux per unit area decreases. This type of trade-off implies that the systems can be optimally designed to achieve the desired objectives.

For this study, the literature reviewed was oriented towards tertiary wastewater applications as well as laboratory and pilot scale tests of osmotic membranes. The literature excluded the extensive basic research on osmotic membranes for other applications (e.g., industrial wastewater treatment). The very extensive patent literature on the different types of membranes, designs, and manufacturing processes was also excluded. Table 120 provides a summary of the current literature pertaining to the performance of reverse osmosis units.

Water Quality Parameters

RO systems are not commonly used specifically for general contaminant removal (with the exception of TDS), as these constituents can be sufficiently removed by other less expensive treatment units. However, there were quite a few references that noted the performance of RO on general contaminant removal from wastewaters as shown in Table 120. The performance of RO systems in terms of percent removal is excellent (90 to 99^+ percent) for all general contaminants except for low molecular components of BOD, or COD, and NO_3.

TABLE 120. LITERATURE REVIEWED PERTAINING TO REVERSE OSMOSIS

Contaminant	Reference Number
Water Quality Parameters	
ABS	534, 547, 901, 1591, 1673
NH_4	90, 1603, 1608, 1609, 1613, 1616, 1636, 1673
BOD	547, 1608
COD	901, 1323, 1603, 1608, 1609, 1613, 1616, 1636, 1673
Chlorides	901, 1603, 1608, 1609, 1629, 1673, 1684
CN	1629, 1684
Fluorides	1608, 1629, 1684
Hardness	547, 901, 1629
NO_3	263, 547, 584, 1608, 1613, 1629, 1673
PO_4	534, 547, 901, 1603, 1608, 1609, 1613, 1629, 1673
SO_4	901, 1603, 1608, 1609, 1629, 1673
TDS	901, 534, 547, 1603, 1608, 1609, 1613, 1636, 1673
Elemental Contaminants	
Aluminum	1608, 1629
Arsenic	1610
Barium	1610
Boron	1608, 1610, 1629, 1684
Cadmium	1323, 1610, 1629
Chromium	1323, 1608, 1610, 1629, 1684
Copper	1323, 1610, 1629, 1684
Iron	901, 1323, 1610, 1629, 1673

TABLE 120. (continued)

Contaminant	Reference Number
Lead	1323
Magnesium	901, 1603, 1608, 1609, 1629, 1670, 1673, 1684
Mercury	1610
Nickel	1323, 1684
Potassium	901, 1608, 1609, 1629, 1670, 1673
Selenium	1610
Silver	1323, 1610
Zinc	1323, 1610
Biocidal Contaminants	
DDT	1591, 1604
DDD	1591, 1604
Aldrin	1604
Organophosphorus insecticide	1604
Chlorinated hydrocarbons	1604
Dieldrin	1604
Herbicides	1604
Lindane	1591, 1604
Pesticides	367, 1604
Biological Contaminants	
Bacteria	1608, 1629
Virus	1608
Synthetic/Organic Contaminants	
Misc. organics	534, 1591, 1608, 1616

Several references discussed the removal of ABS from solution. The presence of highly nonbiodegradable alkyl benzene sulfonate (ABS) even in concentrations of only 1 mg/ℓ can produce undesirable frothing and foams although health implications do not appear significant. Hauck and Sourirajan (547) reported removals of 99^+ percent with a high feed concentration of 300 mg/ℓ . Merten and Bray (901) and an EPA study (1673) both reported ABS rejection of about 98 percent with a variety of membranes. Bennett et al. (46) tested RO performance on a vast array of organics. They showed ABS removals of 90 percent at a 14 gpd/ft^2 flux.

Many studies were conducted that included ammonia removal (see Table 120). Reported removals ranged from 60 to 97 percent. Since ammonia concentrations are not generally of significance in water supplies and NH is readily oxidized in treatment processes, further discussion is not necessary here.

Hauck and Sourirajan (547) studied the performance of RO on hard water and on wastewater. The average BOD removal from secondary effluent was 85.8 and 80.8 percent at 1,000 and 500 psig, respectively. Reference 148 reported BOD secondary effluent BOD rejections varying from 81 to 94 percent with a cellulose acetate membrane.

COD removal is studied more frequently than BOD removal in regard to RO performance, because many of the nonbiodegradable constituents of COD can be effectively removed by RO. A review of all nine references reporting on COD removal shows a removal range of 90 to 99^+ percent with an average of about 95 percent. As a typical example, one of the larger scale pilot programs performed at Hemet and Pomona, California (1323) found that RO provided very good removals of trace organics. The COD of secondary effluent was reduced from 39 mg/ℓ to 1 mg/ℓ at Pomona, while activated carbon effluent COD was reduced from 11.4 mg/ℓ to 0.3-1.0 mg/ℓ . Similar removals were reported from the RO pilot plant at Hemet. It was found that although RO was capable of greatly reducing effluent COD concentrations, the costs for full-scale operation are very high.

Chloride is readily removed by RO. Many studies have evaluated this constituent with removals ranging from 85 to 97 percent (901, 1603, 1608, 1609, 1629, 1673, 1684).

Two references included CN removal in research efforts (1629, 1684). Hindin and Bennett (1629) found that CN removals ranged from 79 to 85 percent at a flux of about 18 gpd/ft^2. A summary report (1684) listed typical CN removal at about 90 percent.

A study by Cruver (1608) reviewed RO performance in removing various contaminants. It was found that fluoride removals ranged from 88 to 98 percent using selective cellulose acetate membranes. Hindin and Bennett (1629) and reference 1684 reported similar results.

RO units are highly efficient at removing hardness (Ca^{++} and Mn^{++}) from water supplies. Hauck and Sourirajan (547) summarized the effects of RO treatment of hard-water supplies in five cities and formal removals ranging from 96 to 99.9 psig as $CaCO_3$. Operating pressure was 1,000 psig with a 90 percent product recovery. Performance was directly related to flux, with the higher removals achieved at lower flow rates. Merten and Bray also reported removals at roughly 99 percent for the more efficient membranes they analyzed.

The reviewed data indicates that nitrate nitrogen is one of the most difficult compounds for RO processes to remove (1673). Hauck and Sourirajan (547) reported NO_3-N removals ranging from 50 to 60 percent. Extensive pilot testing by the EPA at Pomona, California, showed NO_3-N removals of 54 percent over the first 9,475 hours of operation (1603). The five other references showed similar poor removals ranging from 51 to 86 percent and averaging around 60 percent.

Both $PO_4^{=}$ and $SO_4^{=}$, on the other hand, were very readily removed by RO processes. All references reviewed showed removals of 94 percent or better, some up to 99.9 percent.

The primary water treatment use of RO units is to reduce the TDS concentrations of highly mineralized waters. One full-scale example of this is the Orange County Water Districts Water Factory 21 (1636) where 5 mgd of highly treated effluent is given RO treatment prior to recharge of potable aquifers. Results from the operation to date show a 90 to 95 percent TDS removal at 90 percent feed recovery. Results from other references using various membranes, pressures, and fluxes showed that average TDS removals were about 91 percent and ranged from 89 to 99 percent. Removals can be controlled by membrane selection and adjustment of flux. Studies of pilot RO systems at Pomona (1603) showed TOC removals of 86 percent when using primary effluent on the feed.

The major problem areas associated with general contaminant removal are fouling, short membrane life, and associated engineering problems. If the membrane life and flux rate problems are solved, the RO membrane technology may become applicable to wastewater treatment where some sort of reuse is desirable and salt content would restrict reuse. At present, the tubular type membrane systems appear to be more applicable to sanitary wastewaters due to the lower fouling and easier cleaning inherent in this type of design. This advantage may not be as significant in potable water treatment.

Elemental Contaminants

A good deal of available literature on RO membranes deals with removal of elemental contaminants. RO systems can be designed to remove almost any elemental contaminant existing in either an ionic form or colloidal form in water. Generally, multivalent ions (Fe^{+3}, Cu^{++}, Zn^{++}, $SO_4^=$)are rejected more effectively than monovalent ions (B^+, NO_3^-). As previously mentioned, the percentage removal will depend upon the specific membrane and manufacturing procedures. Most references provide a lengthy list of contaminant removals. It would be redundant to discuss each of them here. References 283, 1323, 1625 provide typical summaries of the performance of reverse osmosis units in removing elemental contaminants. Results are shown in Table 121. As shown, RO is generally very effective.

TABLE 121. REVERSE OSMOSIS REMOVAL OF ELEMENTAL CONTAMINANTS (283, 1323, 1629)

	Percent Removal (Single Pass)		
Contaminant	Reference 283	Reference 1323	Reference 1629
Aluminum	-	-	97
Arsenic	90-95	-	-
Barium	90-95	-	-
Boron	-	-	50
Cadmium	90-98	66-98	68-70
Chromium	90-97	82-98	93-98
Fluorides	90-97	-	88-98
Copper	90-97	99	82-96
Lead	90-99	99	-
Iron	90-99	94-99	95-98
Manganese	90-99	-	-
Mercury	90-97	-	-
Nickel	-	98-99	-
Selenium	90-97	-	-
Silver	90-97	96	-
Zinc	90-99	97	-

Biocidal Contaminants

A few studies have been conducted on the removal of biocidals by osmotic membranes (367, 1591, 1604). Excellent removals were reported for a wide variety of pesticides, insecticides, and herbicides including chlorinated hydrocarbons, organophosphorus compounds, and halogenous cyclodienes. However, a considerable amount of this removal can be attributed to adsorption or absorption on the membrane itself. Since these tests were for short time periods relative to commercial application, the long-term rejection may be more complex and may depend upon whether the contaminant is adsorped or absorbed and upon the diffusion rates through the membrane.

With such a large percentage of the removal being related to adsorption/absorption, it is clear that different types of membrane materials will show significantly different results. For example, cellulose acetate (CA) membranes show rather poor performance on the more polar randox and atrazine, whereas the less polar polythylenimine based membranes showed satisfactory performance. Since the active surface on some types of membranes is on the order of a few tenths of a micron, any significant amount of absorption can change the membrane composition and possible rejection of other contaminants.

Reference 367 summarized some results of various studies using cellulose acetate (CA) membranes for the removal of the common pesticide 2, 4-D. Results varied widely from 57 to 99^+ percent removal depending on influent concentrations, esterification, and feed rate. Performance for removal of the insecticide lindane was also sporadic, ranging from rather poor removals up to 84 percent. DDT and DDD have a very low solubility and tend toward micelle formation thus enhancing RO performance. Reported DDT and DDD removals were 97 to 99 percent (367, 1610).

Chian et al. (1604) performed a detailed analysis of the performance of RO units in removing biocidals. Two types of membranes were evaluated, cellulose acetate (CA) and cross-linked polyethylenimine (NS-100). With each membrane, rejection of all types of pesticides (chlorinated hydrocarbons, organophosphorous, and miscellaneous) was better than 99 percent. Specifically, the following chlorinated pesticides were retained at greater than 99 percent: aldrin, lindane, heptachlor, heptachlor epoxide, DDE, DDT, and dieldrin.

Of the organophosphorus pesticides, diazinon was removed at greater than 98 percent, and methylparathion, malathion, and parathion at better than 99 percent. The lowest removals were reported with the CA membrane on randox (72 percent) and atrazine (84 percent). A significant portion of the pesticide removed was adsorbed on the membrane itself: about 80 to 95 percent for hydrocarbons, 30 to 50 percent for organophosphorus compounds, and 5 to 60 percent for the miscellaneous pesticides.

Bennett et al. (1591) reported on the removal of organic refractories including some biocidals. They found that lindane removal was 84 percent; DDT, 99^+ percent; and casein, 99 percent.

The literature data indicate that removal of biocidals is highly variable, depending not only on contaminant concentrations and membrane characteristics, but on synergistic effects of other components in the water. In any case, RO provides a high level of treatment for many of the common pesticides and insecticides.

Synthetic/Organic Contaminants

The performance of RO membranes with respect to removal of synthetic/organic contaminants is similar to that of many of the biocides. In general, these contaminants can be adsorbed, absorbed, rejected or transmitted by the membrane, with removals depending on chemical species and membrane type. In general, larger molecular weight compounds are readily rejected or sorbed by the membrane, whereas low molecular weight compounds (<100) are more likely to pass through.

Studies by Duvel and Helfgott (1616)have shown that the cellulose acetate (CA) type membranes rejection of low molecular wet organics depends upon the molecular weight and molecular size (as determined by steric geometry, with the size being more important) ; it also depends on the ability of the molecule to form hydrogen bonds. The hydrogen bonding behavior affects the solubility in the membrane surface and hence the permeability.

Studies by Hamoda et al. (534) have shown that high flux membranes can be developed that have good rejections (90^+ percent) of tested organic compounds such as sucrose, glutamic acid, starch, sodium stearate, ABS, LAS, and beef extract. Cruver (276) reviewed similar contaminant removals and listed sucrose at 99.9 percent and glucose at 99.5 percent.

Bennett et al. (1591) performed a comprehensive study of the removal of organic refractories by RO. Results showed that the RO process is capable of producing a product water that is extremely low in organic matter in aqueous solution or dispersion, with the exception of those organic compounds that, when in solution or dispersion, have a lower vapor pressure than water.

The classes of compounds that do not appear to be well rejected and are present in wastewater effluents include compounds such as methanol, ethanol, and phenol. No data were found on the low-molecular-weight, halogenated hydrocarbons such as chloroform, bromoform, halogenated ethane, or ethylene compounds; but, based on the present theoretical knowledge, one would expect the removals to be poor.

Biological Contaminants

Due to the large size of biological contaminants - including virus - relative to the effective pore size of RO membranes, high reductions of these contaminants can be expected.

H Hindin and Bennett (1629) conducted microbiological studies to determine the permeation through a porous cellulose acetate membrane of microorganisms found in sewage effluent. Their results showed that *E. coli*, *A. aerogenes*, coliphage T-7 and X-175, and *S. narcesence* were all removed 100 percent by the RO unit, with the exception of one test in which a leak in the membrane may have permitted permeation.

Reference 1662 states that bacterial removal efficiency approximates 99^+ percent when the membrane consists of material that is safe from bacterial decay. Cruver (1608) reports that several studies have shown that 99.9 percent removals of bacteria and virus can be attained.

However, even with these excellent removals, RO processes are not used alone for disinfection because of the presence of imperfections in the membranes. These systems are primarily designed for TDS removal in which small leaks through the membrane and at seal joints are generally inconsequential. Nevertheless, these leaks could be significant when they reduce the removal of virus and bacteria from 99.9999 to 98 percent. To depend upon these systems for 100 percent biological contaminant removal would require continuous monitoring for biologicals and a degree of quality control that would be considered beyond the state of the art for field systems.

The present literature indicates that whereas removals of biological contaminants with RO are very good, they are not as high or as fail safe as other disinfection practices (chlorination, ozonation). It should also be remembered that RO is only a separation process, not a destruction unit, and that the biological contaminants, once removed, will remain in the waste solution.

RECREATIONAL WATER

INTRODUCTION

Public health problems can arise if municipal wastewater effluents are discharged directly into waters used for body contact recreation. In addition, combined sewer overflows as well as storm water runoff from urban areas and animal feedlots can add to the contamination of recreational waters. At a few sites in the United States, land-applied wastewater is being recovered for recreational use (boating, fishing, swimming). Unless properly supervised, such reuse schemes may also pose a hazard to public health (134, 1214).

Nearly all health problems associated with recreational waters can be traced to the presence of pathogenic microorganisms of fecal origin. Human infection is most often the result of direct body contact. As a consequence, recreational water quality standards have been established that identify the maximum allowable concentrations of coliform indicator organisms. The adverse health effects of most other contaminants in recreational waters have received scant attention, since few, if any, outbreaks of disease or infection have been traced to other than biological pathogens. This fact is reflected in the available literature (Table 122).

BIOCIDAL CONTAMINANTS

Pesticides and herbicides found in recreational waters are not considered to pose a significant public health problem. The concentration or dosage of such biocides required to constitute a health hazard is far in excess of water solubility (1455). Four aspects of the problem were addressed:

- Identification of the acute and chronic levels of pesticides toxicity in man;
- The amount of pesticides in the human diet;
- Pesticide concentrations in surface waters; and
- The epidemiology of pesticide exposure.

It was concluded that no significant hazard to public health can be traced to pesticide contact during aquatic recreation.

TABLE 122. LITERATURE REVIEWED PERTAINING TO THE CONTAMINATION OF RECREATIONAL WATERS FROM MUNICIPAL WASTEWATER EFFLUENTS

Contaminant	Reference Number
Water Quality Parameters	
BOD	1342
Chlorides	899
Nitrates	1342
Phosphates	899, 1342
Total dissolved solids	899
Elemental Contaminants	
Boron	899
Biological Contaminants	
Adeno virus	1455
Bacteria	434, 773, 1094
Coliforms	178, 248, 434, 773, 1455
Coxsackie virus (A&B)	1455
ECHO virus	1455
Fecal streptococci	178, 180, 457, 773, 1455
Escherichia coli	180
Hepatitis virus	111
Polio virus	888, 1455
Salmonella	178, 248, 457, 1455
Shigella	178
Virus	178, 773, 888, 900, 1342
Other (general)	180, 279, 434, 1455

BIOLOGICAL CONTAMINANTS

The following types of disease have been associated with primary body contact during aquatic recreation: (1) eye, ear, nose, and throat infections; (2) skin diseases; and (3) gastrointestinal disorders. Enteric infection from recreational water is of far less importance than infection from drinking water, since only small amounts of water are usually consumed during swimming. Nonetheless, cases of typhoid and other intestinal diseases traceable to recreational contact have been reported in the literature, and there is growing concern that viral infections, particularly infectious hepatitis, may be contracted by bathing in sewage-contaminated water (1455).

The magnitude of the health risk posed by bacteriological or viral contamination is uncertain, even when the concentration of specific pathogens is known. When only the indicator organism concentrations are known, the risk is even more difficult to assess. This is because data are lacking from disease outbreaks or epidemiological studies that establish a clear correlation between individual cases of disease or the rate of illness associated with the use of recreational waters, and a given level of any one of the several potential indicators of microbial water quality (831, 1422, 1455).

A number of case reports has linked outbreaks of viral or bacterial disease to swimming in contaminated waters. However, in most cases the occurrence of disease could not be clearly associated with a specific concentration of water quality indicator organisms. Moreover, many infections and toxicities are subclinical in nature, producing only brief or mild illness and going unreported. Most of these diseases can be transmitted by routes other than recreational water use. Consequently, it is difficult to establish a direct statistical relation between recreational water quality, as indicated by the levels of indicator organisms, and the incidence of infectious disease (178). Cabelli et al. (178) have recently embarked on an epidemiological study that is designed to overcome these problems by using improved control populations and a range of indicator organisms. Other authors (434, 1455) have also attempted to develop mathematical models to assess the health risk posed by a particular level of contamination. These models are based on critical assumptions about the probability of infection from a given dose of pathogen and the probability of acquiring such a dose. However, Fuhs (434) admits that there are few accurate estimates of infectious doses of organisms that are of concern in water sanitation. In fact, Envirogenics Co. (1455) used dose-response data derived from non-water-related health programs in their model.

Despite the present lack of precise bacteriological and epidemiological data, authorities have felt obliged to establish standards for recreational water quality. Geldreich (457) argues that the fecal coliform indicator is more reliable than salmonella, fecal streptococcus, or total coliform indicators. He recommends a limit of 200 fecal coliform/100 mℓ for primary contact recreational use.

Pseudomonas aeruginosa has also been proposed as an indicator. This organism is associated with ear infections. It is present in swimming pools and natural swimming areas and is found in human feces. More than half of the illnesses associated with swimming are eye and ear infections, according to a study cited by Cabelli (180). However, no conclusive evidence relates *P. aeruginosa* to swimming-related infections, and the establishment of a reliable standard based on this indicator remains unsupported.

In summary, research has been unable to establish the frequency with which swimming-related infections occur or the hazard associated with a particular level of contamination. The risk of contracting serious or fatal disease appears small, even from waters with poor bacteriological quality. Such severely contaminated beaches have been occasionally used by bathers, without reported ill effects (434). Present standards, although arbitrary, are assumed by authorities to be important in preventing outbreaks of less serious waterborne disease or infection.

FISH

INTRODUCTION

Wastewater and sludge disposal to aquatic environments provides the potential for fish to ingest trace metals, synthetic-organic compounds, biocides, and pathogenic microorganisms present in municipal wastes. This disposal of municipal wastes by direct discharge to surface water bodies is a direct contaminant pathway. There are also less direct pathways, e.g., transport of waste constituents in land runoff, groundwater, or air. In general, the available literature has not been concerned with these less direct pathways, although they may occasionally be significant. For this discussion, attention will be centered on the public health-impairing contaminants that may accumulate in fish to levels hazardous to man. Table 123 is a listing of the literature that was reviewed relevant to fish contamination from wastewater and sludge.

Trace elemental, synthetic-organic, and biocidal contaminants often have the tendency to be bioaccumulated by aquatic organisms, i.e., to be concentrated into their tissues to levels much higher than those in surrounding water. This results from the high solubility of contaminants in lipids relative to their solubility in water. Bioaccumulation is a multistage process: the contaminants are taken into the organism, circulated in the blood, transferred into and out of various organs, and finally metabolized and/or excreted. Under conditions of constant exposure, an organism may eventually reach a quasi-equilibrium in which the concentration of the contaminant in the tissues is constant. It is then possible to define a concentration factor, i.e., the ratio of the concentration of an element or compound in the tissues of an aquatic organism compared to its concentration in the surrounding water under equilibrium or steady state conditions.

Some contaminants tend to be concentrated into the tissues of organisms at levels higher than those in the food ingested by the organisms. This most important phenomenon is termed food chain biomagnification. There is great variation in the feeding habits of different fish species; therefore, the particular feeding habits of each species and the previous steps in its food chain will strongly influence the quantities of contaminants ingested by the species.

TABLE 123. LITERATURE REVIEWED PERTAINING TO FISH

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	1397
BOD	1937
Chlorides	1397, 1398
COD	435
Iodides	647
Phosphates	527
Other (general)	647, 873, 1246
Elemental Contaminants	
Aluminum	995, 1128, 1380, 1390
Antimony	303, 777, 995, 1128, 1380, 1390, 1408
Arsenic	118, 303, 409, 995, 1128, 1380, 1390, 1408
Barium	995, 1128, 1380
Beryllium	995, 1128, 1380
Boron	995, 1113, 1114, 1128, 1380, 1390
Cadmium	118, 207, 303, 365, 472, 539, 859, 873, 995, 1121, 1128, 1247, 1319, 1380, 1390, 1408
Chromium	19, 77, 303, 365, 777, 859, 873, 995, 1121, 1128, 1247, 1307, 1380, 1390, 1408
Cobalt	303, 365, 647, 859, 995, 1121, 1128, 1380, 1390
Copper	19, 77, 274, 365, 472, 504, 505, 712, 777, 859, 873, 995, 1121, 1128, 1380, 1390, 1408

TABLE 123 (continued)

Contaminant	Reference Number
Germanium	995, 1128, 1380, 1390
Iron	19, 274, 647, 995, 1121, 1128, 1380, 1390
Lead	118, 234, 365, 472, 539, 748, 777, 859, 873, 995, 1043, 1045, 1121, 1128, 1380, 1390, 1408
Manganese	274, 365, 600, 647, 995, 1121, 1128, 1380, 1408
Mercury	365, 369, 455, 995, 1128, 1251, 1295, 1380
Molybdenum	303, 995, 1128, 1380, 1390
Nickel	19, 365, 859, 873, 995, 1121, 1128, 1380, 1390, 1408
Selenium	995, 1114, 1128, 1380, 1408
Thorium	995, 1128, 1380
Tin	995, 1128, 1380, 1390, 1408
Uranium	995, 1128, 1380, 1408
Zinc	77, 118, 274, 303, 365, 472, 504, 505, 539, 647, 777, 859, 995, 1121, 1128, 1380, 1390, 1408
Other (general)	303, 365, 647, 873, 893, 1128, 1307, 1380, 1390, 1454, 1456
Biocidal Contaminants	
Chlorinated hydrocarbons	44, 199, 365, 387, 512, 772, 789, 851, 871, 873, 893, 998, 999, 1065, 1163, 1198, 1280, 1294, 1307, 1539, 1561
DDD	425, 429
DDE	199, 421, 429, 851, 972, 1280
DDT	199, 325, 421, 429, 843, 851, 871, 873, 874, 972, 1147, 1198, 1280, 1294, 1307, 1530, 1539

TABLE 123 (continued)

Contaminant	Reference Number
Dieldrin	199, 208, 421, 429, 972, 1280
Endrin	1163
Herbicides	855, 1551
Organophosphorus pesticides	1550
Other (general)	365, 512, 600, 641, 1454, 1456
Synthetic/Organic Contaminants	49, 235, 319, 387, 537, 789, 893, 998, 1163, 1397, 1398, 1564
Biological Contaminants	
Bacteria	131, 1397
Coliforms	1345, 1397, 1458
Fecal streptococci	1397
Parasitic worms	161
Salmonella	161, 1397
Shigella	161, 1345, 1397
Virus	161
Other (general)	161

ELEMENTAL CONTAMINANTS

Lambou and Lim (748) discussed contaminated fish as one of the concentrated sources of lead in man. They found that although lead is relatively insoluble in water, it can be biologically concentrated at all levels of the aquatic food chain. They used limited data from isotopic lead experiments and derived the concentration factors listed below:

	Concentration Factor	
	Bone	Soft tissue
Marine fish	5,400	35
Fresh-water fish	1,400	15

Bioaccumulation of cadmium by bass and bluegill has been observed in the laboratory. For six months, Cearley and Coleman (207) kept largemouth bass and bluegill in aquaria containing various concentrations of cadmium in solution. The fish were exposed to solutions of 805, 80, 8, and 0.5 ppb cadmium, solution was used as a control. The two most highly concentrated solutions were toxic to specimens of both species of fish. The fish accumulated cadmium in concentrations greater than those of the exposure water. The quantity of metal accumulated increased as the exposrue concentration increased. At exposure to 8 ppb cadmium, the maximum total body accumulation by the bass was 8 times greater than that observed in controls; at exposure to 80 ppb cadmium, the accumulation was 15 times greater. At exposure to 8 ppb cadmium, the maximum accumulation by the bluegill was 6 times greater than that observed in controls; at exposure to 80 ppb cadmium, the accumulation was 210 times greater. An equilibrium developed between the cadmium concentrations in the water and in the tissues, as evidenced by the absence of significant additional accumulation after the second month of exposure.

The Illinois River has been receiving municipal and industrial wastes containing trace metals for many years. Mathis and Cummings (859) used atomic absorption spectrophotometry to determine the concentrations of copper, nickel, lead, chromium, lithium, zinc, cobalt, and cadmium in water, sediments, and fish from the river. Ten fish species were examined. Five were carnivorous species, feeding mainly on smaller fish, and five were primarily omnivorous, feeding mainly on insect larvae, molluscs, algae, and aquatic plants. The mean metal concentrations in muscle tissue of these fish are presented in Table 124. All metals were more highly

TABLE 124. RESULTS OF ANALYSES OF VARIANCE AND MEANS OF METAL CONCENTRATIONS IN CARNIVOROUS AND OMNIVOROUS FISH SPECIES FROM THE ILLINOIS RIVER (859)

Metal	Carnivorous Fishes		Omnivorous Fishes		Results of Analysis of Variance*
	Number of Samples Analyzed	Metal Concentration (ppm)	Number of Samples Analyzed	Metal Concentration (ppm)	
Cu	22	0.13	72	0.21	+
Ni	21	0.12	71	0.17	+
Pb	22	0.57	72	0.64	-
Cr	22	0.12	72	0.22	+
Li	15	0.004	58	0.004	-
Zn	22	3.49	72	5.02	+
Co	22	0.10	71	0.10	-
Cd	17	0.03	64	0.03	-

* + indicates significant difference at the 95 percent level;
- indicates no significant difference at the 95 percent level.

concentrated in the fish muscle tissue than in the river water. When carnivorous and omnivorous groups were compared statistically, it was shown that zinc, chromium, nickel, and copper concentrations were significantly higher in the omnivorous than in the carnivorous fish. Concentrations of the eight metals in bottom sediments were considerably higher than in fish muscle. Bottom-dwelling worms and clams closely reflected the concentrations of metals in sediments. A concentration gradient ranging from highest levels in worms, intermediate levels in clams, and lowest levels in fish fillets was observed for copper, nickel, lead, chromium, lithium, cobalt, and cadmium. Zinc was present at highest levels in clams and at lowest levels in fish fillets.

Gavis and Ferguson (455) reviewed the knowledge of the mercury cycle in the aquatic environment. They point out that mercury does not often exist in hazardous concentrations in natural waters; however, in places where mercury compounds have been released to the environment, fish have been able to obtain and concentrate enough mercury to make them a hazardous food source. Furthermore, even after mercury discharge was stopped at these locations, mercury continued to be available from the bottom sediments, where it had been methylated by anaerobic bacteria.

Inorganic mercury in the zero or 2+ oxidation state or methylated mercury can be adsorbed on particles or absorbed by plankton and then ingested by higher organisms. This accumulation of inorganic mercury by organisms may be facilitated by the affinity of zero oxidation state mercury for lipids; however, it is not likely that this is an important factor since the 2+ oxidation state is predominant in oxygenated water where most organisms must live. Forms of methyl mercury, on the other hand, are highly soluble in lipids, facilitating their absorption and bioaccumulation by aquatic organisms (455).

Although available data indicate that algae may be able to bioaccumulate inorganic mercury with concentration factors reaching 100, Gavis and Ferguson could find no data indicating the amount of methylated mercury that organisms are able to bioaccumulate. Further up the food chain the mercury concentrations in organisms increase, indicating that biomagnification occurs. Fish concentrate mercury from food and directly from water. Data were cited which showed that at least 90 percent of the total mercury in contaminated Swedish freshwater and marine fish was usually of the methylated form. Additional data showed that fish accumulated higher mercury concentrations in their tissues as they grew older. None of the literature reviewed by Gavis and Ferguson was concerned with municipal waste discharges (455).

Shin and Krenkel (1251) conducted laboratory studies to determine the uptake rates by fish of methyl mercury biosynthesized in sediments. They found that the higher the mercury concentration in sediment, the more methyl mercury was produced and the greater was the methyl mercury uptake by fish.

Tong et al. (1390) measured the concentrations of 36 trace metals in Lake Cayuga trout aged from 1 to 12 yr. Their studies showed that the concentrations of most metals are not related to fish age. A significant positive correlation with age was found only for chromium, while molybdenum and tin had significant negative correlations. No significant correlations were found for other metals. Mercury concentrations were not determined.

The effects of arsenic in aquatic environments were compared with those of mercury by Ferguson and Gavis (409). Their review indicated that, unlike mercury, arsenic is not concentrated up the food chain. They cited research which showed that arsenic concentrations in organisms are considerably higher than those in the water in which they live. Concentration factors ranging from 2 to 20 have been reported for fresh-water fish. Concentrations in marine plants and animals are generally higher than those in fresh-water species, but evidently little or no food chain biomagnification occurs. The National Academy of Sciences presented average concentration factors for the elements in marine fish muscle. These factors were taken from a large number of references (1128) and are reported for trace metals in Table 125.

The elemental contaminants have been studied by the Southern California Coastal Water Research Project (SCCWRP) as part of an extensive monitoring program of the effects of waste discharges upon Southern California coastal waters (1307). Municipal wastewaters are a major source of trace metals to the Southern California coastal ecosystem. McDermott and Young (873) studied Dover sole living in contaminated sediments around major submarine sewage outfalls in Southern California. They compared the concentrations of seven metals in flesh, gonads, and livers of Dover sole collected around the Palos Verdes sewage outfall with concentrations of the metals in Dover sole collected from a control region in the Santa Barbara channel. Table 126 gives these concentrations as well as the multiplication factors for metal enrichment of the sediments near the outfall relative to the control sediments. In general, no significant metal enrichments were found in the flesh of the outfall specimens compared to the control specimens, although the higher median value for chromium in liver tissue of outfall specimens is significant at the 95 percent level.

TABLE 125. AVERAGE CONCENTRATION FACTORS FOR TRACE METALS IN FISH MUSCLE (1128)

Trace Metal	Concentration Factor
Aluminum	10,000
Arsenic	33,000
Cadmium	1,000
Chromium	70
Cobalt	10
Copper	1,000
Iron	1,600
Manganese	80
Molybdenum	10
Zinc	500

TABLE 126. ESTIMATED SEDIMENT ENRICHMENTS AND MEDIAN CONCENTRATIONS (mg/dry kg) OF SEVEN METALS IN TISSUES FROM DOVER SOLE TRAWLED NEAR THE PALOS VERDES OUTFALLS (STATION T -450) AND FROM A CONTROL REGION IN THE SANTA BARBARA CHANNEL (873)

Trace Metal	Sediments Outfall: Control	Flesh		Gonads		Liver	
		Outfall	Control	Outfall	Control	Outfall	Control
Cadmium	150	<3.0	<3.0	<3.0	<3.0	3.8	6.4
Chromium	16	<0.2	<0.2	0.4	<0.2	0.4	<0.2
Copper	20	0.6	0.5	10.7	12.7	7.4	9.5
Nickel	2.4	<2.0	<2.0	<4.8	<2.0	2.6	1.4
Lead	52	1.3	<1.1	<1.0	<1.0	5.0	4.6
Zinc	12	39.4	38.0	214	233	91.8	108

Dover sole was also selected by de Goeij et al. for a study of trace element concentrations in fish samples from areas down current from and around the major Los Angeles sewage outfalls (303). Neutron activation analysis of Dover sole liver tissue and bottom sediment from both contaminated and uncontaminated areas was performed. No significant differences were observed in the levels of 11 trace elements between the two samples, although sole is a "bottom feeder" fish, and the ocean sediments in the areas around the sewage outfalls had trace metal concentrations sometimes 10 to 150 times higher than in uncontaminated areas. Table 127 gives the measured trace element concentrations in liver tissue and 95 percent confidence levels for samples from an area with only natural sediment trace metal accumulation and from areas near sewage outfalls with intermediate to very high trace metal accumulation. These data indicate that the concentrations of 11 metals in livers of Dover sole did not increase as a result of exposure to and feeding in contaminated sediments.

In the laboratory, Sherwood and Wright (1247) exposed another type of marine flatfish, the speckled sanddab, to dissolved hexavalent chromium and measured the bioaccumulation of the element. The fish were exposed to 16, 95, 550, and 3,000 ppb of dissolved potassium dichromate for 44 days. These levels are above the range of common levels for world ocean waters (0.05 to 0.5 ppb) and are above the level for water in the immediate vicinity of a Southern California wastewater outfall. The latter would average 5.0 ppb if the wastewater were diluted 100:1 immediately upon discharge. During the experiment, even at the lowest exposure concentration, there was significant proportional accumulation of chromium in both external and internal tissues: intestine, skin, liver, and muscle. These tests confirmed the biological availability of low concentrations of dissolved hexavalent chromium to fish. Sherwood and Wright (2147) also exposed speckled sanddabs to precipitated trivalent chromium in the form of chronic chloride. Hydroxide precipitate was produced, added to test tanks, and allowed to settle in a layer on the bottom. Seawater was added on a flow-through basis. Bioaccumulation did not occur in skin, muscle, or liver, suggesting that the trivalent chromium precipitate was not biologically available.

In summary, the information reviewed appears contradictory. High concentration factors indicate a tendency for significant bioaccumulation of metals at all levels of the food chain. However, the environmental significance of measured concentration factors, as discussed by the National Academy of Sciences (1128), is difficult to establish. These factors are often determined in the laboratory, eliminating the complex

TABLE 127. TRACE ELEMENT CONCENTRATIONS, SHOWN TO A 95% CONFIDENCE LEVEL, IN DOVER SOLE LIVER COLLECTED FROM LOCATIONS WITH DIFFERENT DEGREES OF SEDIMENT CONTAMINATION (303)*

	Degree of Sediment Contamination with Trace Metals			
Trace element	Background 6 samples	Intermediate 9 samples	High 12 samples	Very high 12 samples
Antimony	0.0035 ± 0.0016	0.0028 ± 0.0009	0.0022 ± 0.0006	0.0029 ± 0.0009
Arsenic	3.1 ± 0.7	1.3 ± 0.2	1.5 ± 0.3	1.3 ± 0.2
Cadmium	0.58 ± 0.29	0.29 ± 0.06	0.33 ± 0.16	0.19 ± 0.06
Cobalt	0.061 ± 0.016	0.041 ± 0.006	0.043 ± 0.012	0.037 ± 0.008
Copper	2.2 ± 0.5	2.5 ± 1.0	3.4 ± 1.2	2.0 ± 0.4
Iron	182 ± 59	194 ± 31	176 ± 35	209 ± 27
Mercury	0.11 ± 0.04	0.14 ± 0.03	0.19 ± 0.06	0.11 ± 0.02
Molybdenum	0.13 ± 0.04	0.093 ± 0.018	0.081 ± 0.011	0.081 ± 0.017
Selenium	1.2 ± 0.3	0.66 ± 0.19	0.97 ± 0.25	0.65 ± 0.15
Silver	2.2	2.1 ± 0.7	6.3 ± 2.2	1.8 ± 0.5
Zinc	27 ± 4	20 ± 5	25 ± 4	26 ± 3

* All concentrations are given in microgram element/gram wet material, except for silver which is presented in arbitrary units.

set of relationships that determines the total availability of a trace element to an organism. Few studies have tried to identify municipal waste as the source of contaminants to fish. Studies by the SCCWRP (1307), and by de Goeij et al. (303) are fairly unique. The studies show no significant increase in the metal concentrations of fish that live and feed in contaminated sediments near wastewater outfalls.

BIOCIDAL AND SYNTHETIC/ORGANIC CONTAMINANTS

Polychlorinated byphenyls (PCB's) are an important contaminant because of their potentially serious public health effects and ability to concentrate heavily in fish tissue. During the period from 1973 to 1975, the Food and Drug Administration found that 3 percent of the interstate commerce fish tested exceeded the PCB limit of 5 ppm. All of the contaminated fish were from the Great Lakes area (1163). Severe contamination of fish by PCB's in the Hudson River has also been reported (387). Over 80 percent of the fish sampled at one point on the river contained more than 5 ppm PCB's (a rock bass reportedly contained 350 ppm PCB's). In the past, industrial effluent discharges have been responsible for much of the PCB loading to aquatic systems; however, as restrictions on these discharges are more rigidly enforced, a greater percentage of PCB's discharged to water bodies may be contained in municipal wastes. According to Nisbet (998), PCB's have a high tendency for bioaccumulation because of their high solubility in lipids. Within fish, PCB's are further concentrated into certain organs, especially those with high lipid content. Nisbet further points out that fewer laboratory experiments and more field measurements are needed to quantify the degrees of bioaccumulation that can be expected in different circumstances. They cite numerous studies which showed that under laboratory conditions, fish bioaccumulate PCB's to whole-body concentrations of 30,000 to 300,000 times higher than those in surrounding water. Concentration factors of up to two million were measured for lipids.

Chronic exposure of fish to Aroclor 1061 was investigated by Hansen et al. (537). In a 42-day experiment, when fish were exposed to concentrations of 32 and 100 mg/ℓ, mortality began in the second week. The fish that died exhibited symptoms of poisoning and showed severe liver and pancreatic alterations. The maximum whole-body residue (wet wt) of PCB was 17,000 times the nominal PCB concentration in test water. The actual body burden for toxicity was several percent PCB by weight. When placed in PCB-free water, the fish were able to remove some of the PCB from their tissues and after 56 days showed an average reduction of 51 percent in PCB concentration.

Lieb et al. (789) reported PCB accumulation in rainbow trout that were fed Aroclor 1254 at 15 ppm. The total retention of PCB's from the diet was 68 percent for a 32-week feeding period. The PCB whole-body concentration stabilized at 8.2 ppm during that period. PCB did not appear to be eliminated after exposure ceased, even when the fish were starved. The fish did not appear to be adversely affected by the PCB.

The results of the above-mentioned laboratory studies can be compared to results from studies of the natural aquatic environment. Concentrations of PCB in Cayuga lake trout reported by Bache et al. (49) are listed in Table 128. PCB concentrations were found to be strongly related to fish age, with higher concentrations being observed in older fish.

Plankton are an important link in the introduction of PCB's into the aquatic food chain. Direct surface adsorption of PCB's by plankton allows these substances to remain in the pelagic environment for long periods of time. Linko et al. (804) measured PCB concentrations in sediment and plankton from the southwestern coast of Finland. They found the concentrations in sediment samples to be lower than those in plankton samples.

Greichus et al. (512) measured organochlorine insecticide and PCB concentrations in water, bottom sediments, and fish at Lake Poinsett, South Dakota. As shown in Table 129, insecticide concentrations in bottom sediments are about 7 times higher than those in the water column, and concentrations in the fish range from 10 to 60 times higher than those in the bottom sediments. The levels of PCB are higher than those of insecticides in both sediments and fish.

The persistence of DDT and its metabolites in aquatic ecosystems and the tendency for food chain biomagnification was investigated by Dimond et al. (325). The effect of a single application of DDT to forestland on trout in streams was studied for 10 yr. Upon application of DDT, the mean concentration in trout rose to 8.21 ppm; 1 yr later, it decreased to 3.51 ppm; and after 2 yr, it was 2.10 ppm. Further reduction in DDT levels was gradual. Even after 10 yr, concentrations were still from 3 to 9 times higher than those in streams never contaminated with DDT. The average concentration after 10 yr was 0.35 ppm in treated streams, compared to an average of .07 ppm in untreated streams.

Laboratory studies by Chadwick and Brocksen (208) indicate that food organisms are a much less important source of the pesticide dieldrin to aquatic organisms than water. Dieldrin in solution appears to be immediately and directly

TABLE 128. RESIDUES OF PCB's IN CAYUGA LAKE TROUT AS A FUNCTION OF MATURITY; J. JUVENILE; M. MALE; F. FEMALE (49)

Age (yrs)	Sex	Length (cm)	Weight (g)	PCB (ppm)
1	J			0.6
1	J			1.6
1	J			0.5
1	J			1.2
2	J	27.7	181	2.0
2	J	28.7	226	1.3
2	J	33.5	407	2.5
3	J	44.5	815	2.2
3	J	44.5	725	2.4
3	J	41.1	770	1.2
4	J	53.8	1310	3.5
4	J	50.3	1160	4.1
4	J	55.1	1359	5.1
5	M	61.0	2030	5.7
6	M	63.5	2440	3.4
6	M	66.4	2850	9.7
6	F	68.3	2310	8.6
7	M	63.5	2260	4.0
7	M	68.9	3300	5.5
7		59.7	1990	10.0
8	F	75.2	3390	17.5
8	M	71.6	2805	13.4
8	F	69.0	3300	4.5
9	F	71.2	3390	30.4
11	M	80.3	4200	12.4
12	M	71.6	2535	13.4
12	M	75.5	3120	26.2
12	F	70.6	3440	7.4

TABLE 129. LEVELS OF PCB's AND INSECTICIDES IN FISH AND BOTTOM SEDIMENTS (512)

Material Sampled	No. of Samples	Avg. Total Insecticides ppm	Average PCB's ppm
Water	6	0.00023	<0.0005
Bottom sediments*	6	0.0016	0.0064
Fish**			
Carp	2	0.092	0.11
Crappie	2	0.016	<0.05
Buffalo	2	0.032	0.06
Bullhead	2	0.031	0.11

* Dry wt basis.
** Each sample is a composite of 10 fish (wet wt basis).

available for accumulation in aquatic animals; however, accumulation of dieldrin via food organisms may be slower since food is consumed in smaller amounts.

Frank et al. (429) report data on organochlorine insecticide residues in fish tissue at four study areas in southern Ontario, Canada, which had received different quantities of DDT and dieldrin. The concentrations of dieldrin, and DDT and its metabolites in the fish tended to depend on the feeding habits, fat content, and age of the fish. Low total DDT concentrations (0.1 to 1.0 ppm) were found in low-fat carnivores; higher concentrations (1.0 to 10 ppm) were found in high-fat bottom and plankton feeders; and the highest concentrations (over 10 ppm) were found in high-fat carnivores. A general trend of increasing tissue concentrations with increasing body weight was also noted.

Yu et al. (1551) investigated the fate of the herbicide, dicamba, in a model aquatic ecosystem. They found no evidence that dicamba and its metabolites are biomagnified in the food chain.

SCCWRP demonstrated that the discharge of chlorinated hydrocarbons in municipal wastewaters resulted in increased concentrations of the chlorinated hydrocarbons in sediments around Southern California sewage outfalls. A close correspondence between DDT and PCB concentrations in fish and in sediments of the outfall areas has been observed (365).

SCCWRP collected Dover sole from around Southern California's five major submarine sewage outfall systems and at control areas off Dana Point and Santa Catalina Island (871). Concentrations of DDT and PCB compounds in fish muscle tissue were measured in 1971-72 and 1974-75, as shown in Table 130. The highest PCB concentrations were in specimens collected near the Palos Verdes, Santa Monica, and Orange County submarine sewage outfalls. These concentrations were significantly higher than those found at the control areas.

The major municipal wastewater discharge point for DDT is at Palos Verdes. Approximately 75 percent of the muscle tissue samples from Dover sole caught in 1971 and 1972 near Palos Verdes contained over 5 ppm DDT (1530). As shown in Table 130 , the median concentration was 9.0 ppm in 1971-72 and 12 ppm in 1974-75, despite an 80 percent reduction between 1971 and 1973 in the amount of DDT discharged annually at Palos Verdes.

DDT concentrations in the flesh of bottom-feeding fish - such as black perch and kelp bass caught off Palos Verdes

TABLE 130. CHLORINATED HYDROCARBON CONCENTRATIONS IN MUSCLE TISSUE OF DOVER SOLE (871) (mg/wet kg)

Region	1971-72 Median (Range)	1974-75 Median (Range)	1971-72 Median (Range)	1974-75 Median (Range)
Pt. Hueneme	0.4 (0.3-0.4)	0.1 (0.1)	0.1 (0.1)	0.06 (0.05-0.07)
Santa Monica	1.5 (0.9-7.2)	1.4 (0.6-1.7)	1.5 (0.4-2.8)	2.0 (1.0-2.3)
Palos Verdes	9.0 (3.2-45)	12 (9.7-25)	1.9 (0.7-6.6)	1.3 (0.06-2.2)
Orange Co.	0.8 (0.7-4.4)	3.6 (0.7-26)	0.8 (0.2-1.2)	0.6 (0.3-2.6)
Dana Pt.	0.2 (0.2-0.3)	0.2 (0.1-2.7)	0.06 (0.03-0.09)	0.2 (0.05-0.6)
San Diego	-- --	0.05 (0.04-0.3)	-- --	0.2 (0.08-0.4)
Santa Catalina Island	0.1 (0.1-0.2)	0.1 (0.1)	0.04 (0.04)	0.05 (0.03-0.07)

in 1973 - ranged from 0.82 to 87 mg/wet kg (black perch) and from 0.37 to 29 mg/wet kg (kelp bass). Concentrations in approximately half of all specimens exceeded 5 mg/wet kg. By contrast, a surface feeding predator, the bonita, had a median DDT concentration of 15 to 500 times lower than the median concentrations found in the bottom feeders (1530).

BIOLOGICAL CONTAMINANTS

Many of the biological contaminants, when ingested by fish, may survive in fish viscera, since fish do not have a permanent coliform or streptococcus flora in their intestinal tracts. The population of microorganisms in fish is a reflection of the population in the external environment. Bacteria that are indicators of human fecal contamination, including *Streptococcus faecalis* and *Salmonella typhimurium*, have been found in the intestinal tract of various species of fresh-water fish (1397).

The intestinal contents of 78 fish from a moderately polluted river were examined in a study cited by Tsai (1397). The fecal coliform densities were lowest in bluegill sunfish (less than 20/g) and highest in catfish (1,090,000/g); levels of fecal streptococci for the two species were 200 and 240,000/g, respectively. Apparently, these differences were due mainly to habitats and feeding habits of the species. In another study cited by Tsai, carp were artificially infected with typhoid bacteria. The bacteria were present in all organs from 24 to 48 hours after infection and persisted for 8 to 14 days; they did not disappear until after four to six weeks, although the fish remained healthy.

Temperatures attained by cooking are not, in general, high enough to destroy typhoid bacteria; therefore, it is theoretically possible for bacteria to be transmitted to humans. Research cited by Tsai showed that fish taken from heavily polluted streams contained specific antibodies to the bacteria that cause human pseudotuberculosis, paratyphoid fever, bacillary dysentery, and a variety of chronic infections. Fish taken from mildly polluted streams were free of such antibodies.

Shigella and enteropathogenic *Escherichia coli* were isolated in fish from contaminated river water (161). Certain parasitic diseases are maintained by the cycle: human feces to fresh water; water to copepods or snails; and copepods or snails to fish. *Clinorchis sinensis*, *Diphyllobothrium latum*, and *Echinostoma ilocanum* are human parasites with such life cycles. An outbreak of diphyllobothriasis was reported from eating fish caught in sewage-polluted water.

The research indicated that fish living in contaminated water can be vectors of pathogenic bacteria to man. However, this has not been a health problem in the United States because fish are customarily cleaned (removal of viscera) and cooked before eating. Obviously, improper cleaning or storage of fish, which allows pathogenic organisms to enter the flesh and/or multiply, may create health problems.

SHELLFISH

INTRODUCTION

Filter-feeding shellfish (oysters, clams, etc.) are a major potential pathway of contaminants from wastewater to man. Such shellfish filter large volumes of water through their bodies to obtain food and oxygen. In this process they retain suspended particulate matter. A significant portion of the biological, trace-elemental, synthetic-organic, and biocidal contaminants discharged to water bodies in municipal wastes are associated with the particulate phase, so there is the potential for shellfish to effectively concentrate these contaminants.

While there is relatively little research directly linking shellfish contamination with municipal wastewater discharges, the evidence is clear that shellfish taken from contaminated waters will be contaminated. Table 131 is a listing of the literature that was reviewed relevant to shellfish contamination from wastewater and/or wastewater sludges.

ELEMENTAL CONTAMINANTS

Pringle et al. (1121) reported the ability of estuarine shellfish to concentrate trace elements to levels many hundreds of times the levels found in estuarine water and seawater. Average concentrations of trace metals in shellfish taken from Atlantic coastal waters are shown in Table 132. Species differences in the amount of bioaccumulation are evident. The averages in Table 132 reflect shellfish accumulation of metals at hundreds of locations along the Atlantic Coast of the United States, with widely varying degrees of pollution.

Table 133 shows the range of metal concentrations in samples from Atlantic and Pacific waters. The varieties of environmental contamination at different locations are a factor contributing to the broad range of metal concentrations in shellfish shown in Table 133.

A survey for the U.S. Environmental Protection Agency (EPA) (1241) of shellfish beds in San Francisco Bay reported metal concentrations in shellfish similar to those reported by

TABLE 131. LITERATURE REVIEWED PERTAINING TO SHELLFISH

Contaminant	Reference Number
Water Quality Parameters	
BOD	226
Iodides	647
Other (general)	226, 435, 828, 919, 1459
Elemental Contaminants	
Aluminum	995, 1128, 1380
Antimony	995, 1128, 1380
Arsenic	409, 995, 1128, 1380
Barium	995, 1128, 1380
Beryllium	995, 1128, 1380
Boron	995, 1113, 1128, 1380
Cadmium	135, 136, 162, 365, 502, 539, 698, 699, 732, 828, 995, 1128, 1135, 1241, 1378, 1380
Chromium	162, 365, 502, 611, 698, 699, 732, 828, 859, 995, 1128, 1137, 1307, 1380, 1537, 1540
Cobalt	162, 365, 647, 859, 995, 1128, 1380, 1538
Copper	135, 136, 162, 365, 502, 611, 698, 699, 732, 828, 995, 1128, 1135, 1137, 1307, 1380, 1537, 1540
Germanium	995, 1128, 1380
Iron	55, 135, 136, 162, 365, 611, 647, 995, 1128
Lead	88, 135, 136, 162, 365, 502, 539, 611, 698, 732, 828, 859, 995, 1128, 1241, 1537, 1540

TABLE 131 (continued)

Contaminant	Reference Number
Manganese	136, 365, 502, 600, 647, 828, 853, 995, 1128, 1380, 1538
Mercury	285, 369, 370, 431, 437, 625, 731, 995, 1128, 1264, 1295, 1380
Molybdenum	995, 1128, 1380
Nickel	135, 162, 267, 365, 698, 699, 828, 859, 995, 1128, 1537
Selenium	995, 1114, 1128, 1380
Thorium	995, 1128, 1380
Tin	995, 1128, 1380
Uranium	995, 1128, 1380
Zinc	135, 162, 365, 502, 539, 611, 647, 698, 732, 828, 859, 995, 1128, 1135, 1380, 1537, 1538, 1540
Other (general)	647, 828, 859, 1121, 1241, 1380, 1454, 1456, 1537
Biocidal Contaminants	
Chlorinated hydrocarbons	173, 432, 537, 554, 804, 851, 997, 998, 1065, 1146, 1163, 1197, 1438, 1529, 1531, 1532
DDD	346, 422, 1408
DDE	316, 422, 713, 1408
DDT	346, 422, 554, 654, 713, 798, 922, 1044, 1045, 1074, 1306, 1307, 1390, 1408, 1446, 1525
Dieldrin	421, 422, 1081
Endrin	173, 1163
Herbicides	806, 1197

TABLE 131 (continued)

Contaminant	Reference Number
Other (general)	600, 806, 1197, 1454, 1456
Synthetic/Organic Contaminants	319, 402, 570, 1306, 1307, 1564
Biological Contaminants	
Bacteria	161, 238
Coliforms	177, 973, 1241, 1307
Coxsackie virus (A & B)	905
ECHO virus	905
Parasitic worms	161
Polio virus	322, 323, 905
Salmonella	161, 1241
Shigella	161
Virus	161, 177, 238, 322, 468, 973, 1307
Other (general)	161

TABLE 132. AVERAGE TRACE METAL CONCENTRATIONS IN SHELLFISH TAKEN FROM ATLANTIC COAST WATERS IN PPM (WET WT)(1121)

Element	Eastern Oyster	Soft Shell Clam	Northern Quahaug
Zinc	1428	17	20.6
Copper	91.50	5.80	2.6
Manganese	4.30	6.70	5.8
Iron	67.00	405	30
Lead	0.47	0.70	0.52
Cobalt	0.10	0.10	0.20
Nickel	0.19	0.27	0.24
Chromium	0.40	0.52	0.31
Cadmium	3.10	0.27	0.19

TABLE 133. RANGES OF TRACE METAL CONCENTRATIONS IN SHELLFISH HARVESTED FROM ATLANTIC AND PACIFIC IN PPM (WET WT) (1121)

Element	Oysters		Soft Shell Clam	Northern Quahaug
	East Coast, U.S.	West Coast, U.S.		
Zinc	180 -4120	86 -344	9.0 - 28	11.50-40.20
Copper	7.0 - 517	7.80- 37.50	1.20- 90	1.0 -16.50
Manganese	0.14- 15.0	0.90- 16	0.10- 29.90	0.7 -29.70
Iron	31 - 238	15.30- 91.40	49.70-1710	9.0 -83.0
Lead	0.10- 2.30	0.10- 4.50	0.10- 10.20	0.10- 7.50
Chromium	0.04- 3.40	0.10- 0.30	0.10- 5.0	0.19- 5.80
Nickel	0.08- 1.80	0.10- 0.20	0.10- 2.30	0.10- 2.40
Cobalt	0.06- 0.20	0.10- 0.20	0.10- 0.20	0.10- 0.20
Cadmium	0.10- 7.80	0.20- 2.10	0.10- 0.90	0.10- 0.73

Pringle et al. (1121). The data represents 16 sampling stations. The ranges of trace metal concentrations are as follows:

Element	Concentration (mg/kg wet)
Cadmium	0.06 to 1.41
Chromium	0.10 to 6.63
Copper	1.12 to 48.2
Lead	0.23 to 18.7
Mercury	0.01 to 0.15
Zinc	8.48 to 157

Cadmium in shellfish meat from two stations exceeded 0.5 ppm (0.58 ppm and 1.41 ppm), and lead concentrations exceeded 7 ppm. All levels of mercury from the specimens, however, were below 0.5 ppm. Zinc concentrations were high in oysters, as was found earlier by Pringle et al. (1121).

Mathis and Cummings (859) analyzed fresh-water clams taken from the Illinois River, which has been receiving municipal and industrial wastes containing trace metals for many years. Metal concentrations in the clams (Table 134) reflected the levels found in the river bottom sediments (Table 135); concentrations in the clams exceeded levels found in the water, but were similar to or less than the concentrations in the sediments.

The uptake of mercury by shellfish depends upon the chemical and physical form of the metal. The uptake of mercury and its compounds has received a great deal of attention as a consequence of the Minamata Bay incident in Japan. Irukayama (625) reported that abundant inorganic water-soluble mercury compounds had accumulated in the mud of Minamata Bay, although these compounds were scarcely accumulated by shellfish cultured experimentally in seawater containing the mud. Further experimentation led to the conclusion that an organo-mercury compound -- probably methyl or ethyl mercury -- dissolved in the seawater of the bay had accumulated directly in fish and shellfish. This compound was finally identified as methyl mercury chloride, a somewhat water-soluble compound that had been discharged to the bay in sludge from an acetaldehyde manufacturing plant. After the discharge from the plant was discontinued, the mercury content in shellfish decreased in two years from 85 to 10 ppm, although abundant inorganic mercury compounds were still present in the sediment.

Further research has indicated that any form of mercury that is discharged to water bodies may be converted to methyl

TABLE 134. CONCENTRATION OF METALS IN CLAMS
(859)

Fusconaia flava

Metal	No. of Samples Analyzed	Concentration Mean (ppm)	Concentration Range (ppm)
Copper	17	1.7	0.9 to 2.0
Nickel	17	2.1	0.7 to 3.0
Lead	17	3.7	1.8 to 5.1
Chromium	17	7.7	1.1 to 11.6
Lithium	17	0.109	0.042 to 0.190
Zinc	17	66	25 to 120
Cobalt	17	1.2	0.6 to 1.6
Cadmium	17	0.69	0.36 to 1.17

Amblema plicata

Metal	No. of Samples Analyzed	Concentration Mean (ppm)	Concentration Range (ppm)
Copper	25	1.2	0.3 to 3.2
Nickel	25	1.1	0.4 to 2.3
Lead	25	2.7	1.1 to 7.6
Chromium	25	4.4	0.6 to 9.9
Lithium	25	0.076	0.0005 to 0.260
Zinc	23	95	40 to 178
Cobalt	25	0.7	0.4 to 1.2
Cadmium	25	0.38	0.15 to 1.41

Quadrula quadrula

Metal	No. of Samples Analyzed	Concentration Mean (ppm)	Concentration Range (ppm)
Copper	20	1.7	1.1 to 3.6
Nickel	20	0.9	0.4 to 1.6
Lead	20	2.2	0.9 to 3.8
Chromium	20	4.7	1.8 to 8.3
Lithium	20	0.063	0.036 to 0.100
Zinc	19	48	28 to 64
Cobalt	20	0.8	0.5 to 1.3
Cadmium	20	0.56	0.31 to 1.37

TABLE 135. MEANS AND RANGES OF METAL CONCENTRATIONS IN BOTTOM SEDIMENTS OF THE ILLINOIS RIVER AND THREE NONINDUSTRIAL-USE STREAMS (859)

Metal	Concentration (ppm)			
	Illinois River		Three Nonindustrial-Use Streams	
	Mean	Range	Mean	Range
Copper	19	1 to 82	7.7	3.5 to 11.2
Nickel	27	3 to 124	16	10 to 22
Lead	28	3 to 140	17	13 to 27
Chromium	17	2 to 87	6	3 to 7
Lithium	3.8	0.5 to 16.3	3.8	1.7 to 5.7
Zinc	81	6 to 339	30	18 to 41
Cobalt	6	1 to 18	6	4 to 8
Cadmium	2.0	0.2 to 12.1	0.4	0.3 to 0.5

mercury by microorganisms living in bottom sediments (437, 731). Metallic, inorganic, and phenyl mercury can thus be made available for accumulation in shellfish. Kopfler (731) exposed oysters to different seawater solutions containing mercury as (1) inorganic mercuric chloride, (2) methyl mercuric chloride, and (3) phenyl mercuric acetate. When exposed to 1 ppb mercury in any of these three forms, oysters rapidly concentrated mercury in their tissues far in excess of the 0.5 ppm limit recommended by the U.S. Food and Drug Administration (FDA).

Los Angeles County's submarine discharge of municipal wastewater off Palos Verdes Peninsula is the single largest man-related source of trace metals to Southern California coastal waters. Bottom sediments around the outfall system are highly contaminated by a number of trace metals. Young and Jan (1537) reported abnormal levels of seven metals in three tissues of filter-feeding rock scallops (Hinites multirugosus) that were collected in the discharge zone and thus had been exposed to suspended wastewater particulates. The trace metal concentrations in these scallops were compared with concentrations from control regions. The "contamination ratios" (outfall-to-control region ratios of mean metal concentrations in scallops) are presented in Table 136. This data shows that rock scallops living in the wastewater discharge area accumulated trace metals above normal levels; 16 of the 19 contamination ratios are greater than 1.0.

TABLE 136. "CONTAMINATION RATIOS" (OUTFALL-TO-CONTROL REGION RATIOS OF MEAN METAL CONCENTRATIONS) FOR SEVEN METALS IN THREE SCALLOP TISSUES (1537)

Metal	Digestive Gland	Gonad	Adductor Muscle
Silver	7.4	4.4	3.2
Cadmium	0.93	0.48	2.8
Chromium	19	6.7	7.0
Copper	3.0	1.4	2.6
Nickel	0.87	2.8	1.8
Lead	3.0	--	--
Zinc	1.3	2.3	1.1

The average mercury concentrations in tissues of mussels from Palos Verdes were reported by Eganhouse and Young (370). Concentrations in digestive gland tissues, adductor muscle, and gonadal tissues were 62.8, 61.8, and 14.2 μg/wet kg, respectively -- two to three times higher than levels in mussels from control regions. The high values appeared to be related to the sewage outfall. Experiments were performed to determine the availability of mercury discharged in the Palos Verdes effluent to mussels (Mytilus californianus) (370). Mussels from an area of relatively low mercury levels were suspended near the wastewater outfall and monitored for mercury uptake as a function of time. Of three tissues analyzed, digestive gland tissues accumulated mercury at the fastest rate and contained the highest levels. Adductor muscle and gonadal tissues had markedly slower accumulation rates.

The mercury concentrations in various tissues of crabs, prawns, snails, urchins, and sea slugs from the Palos Verdes shelf measured in 1975 (369) were quite low, in no case approaching 0.5 ppm (FDA guideline for edible seafood). Organic mercury constituted the major fraction of total mercury in muscle tissues of several of the animals. An analysis of the data with respect to proximity to the wastewater outfall showed no distinct pattern. Thus, the investigators concluded that the tissue mercury content of these benthic animals was not governed significantly by sediment concentrations.

Similarly no significant correlations were found between concentrations of 12 trace metals in bottom sediments and in six shellfish species growing at five locations in the San Francisco Bay estuary. Mercury concentrations in two harvestable shellfish species -- Mya arenaria and Tapes japonica -- at all sampling locations were below 0.5 ppm, wet weight. Cadmium concentrations in both species from several locations reached 3 ppm, dry weight, and lead levels at several sites ranged between 12 and 19 ppm. Metal concentrations in oysters (Crassostrea gigas) from a relatively contaminated area inside San Francisco Bay and a relatively clean area outside the bay were dramatically different. For example, the heptopancreas of bay oysters had concentrations of silver, cadmium, copper, mercury, and zinc that were respectively 103, 8, 31, 17, and 22 times higher than the concentrations observed in the hepatopancreas of clean-water oysters.

SCCWRP (1307) found that submarine discharges of wastewater near two of California's harbors did not appear to have caused increased copper levels in intertidal mussels (Mytilus edulis) growing at the bases of these outfalls; vessel antifouling paints evidently caused increased copper levels in mussels within the harbors. A third wastewater outfall off Palos Verdes Peninsula near San Pedro harbor did seem to cause increased copper concentrations in mussels at the base of this outfall system; the mussels contained an average of 47 ppm copper compared to the 20 ppm of the control mussels.

SYNTHETIC/ORGANIC AND BIOCIDAL CONTAMINANTS

The synthetic-organic or biocidal compounds most often studied as shellfish contaminants are DDT and PCB's. Heesen and McDermott (554) reported that relatively large concentrations of DDT were found in a variety of shellfish collected in coastal waters of Los Angeles and Orange Counties in California. They measured concentrations of chlorinated hydrocarbons in the market crab (Cancer anthonyi), which feeds on bottom material that can be significantly contaminated with trace constituents. The results indicated that DDT and PCB concentrations in the flesh of crabs collected around major submarine sewage outfalls were not above 5 ppm. Concentrations of DDT were several times higher in crabs around sewage outfalls than in those around other coastal areas; however, PCB concentrations were similar.

Studies performed by SCCWRP (1306) suggest that intertidal mussels (Mytilus californianus) rapidly reflect changes in the seawater concentrations of chlorinated hydrocarbons by changes in their own tissue concentrations. A 1971 survey of the Los Angeles Bight (1531) revealed a distinct correlation between the concentrations of both DDT and PCB 1254 in mussels and the proximity of the station from which the mussels were taken to the Palos Verdes submarine sewage outfall system, a major source of DDT and PCB's to the region. For over 100 km in most directions from the discharge area, DDT concentrations above baseline levels were detectable; a maximum concentration of 4,200 ppb total DDT was found in mussels near the discharge. In comparison, the median concentration at six control stations was only 70 ppb.

Since 1971, there has been significant reduction in the concentrations of these chlorinated hydrocarbons in municipal wastewater effluent discharged off Palos Verdes. The input of total DDT compounds to the Bight decreased by approximately 90 percent. Decreases in DDT and PCB concentrations in mussels sampled from a beach at the base of the outfall system between 1971 and 1973 were similar to concentration decreases in the effluent. A follow-up experiment was performed that similarly showed elimination of DDT metabolite from mussel tissue when exposure to contaminated water ceased.

Although greatly reduced, the influence of the wastewater discharge was still evident in 1974 (1307); in Palos Verdes mussels, DDT concentrations and PCB 1254 concentrations were 10 and 100 times above baseline values. Both these studies (1306, 1323) demonstrated the usefulness of the mussel as an indicator of chlorinated hydrocarbon contamination of nearshore waters. As a result, an offshore caged mussel biomonitoring program was set up (1307). The results of the program, as expected, showed a direct relationship between uptake of DDT and PCB and proximity to contaminated bottom sediments or a wastewater plume. Specimens suspended near the bottom became approximately 10 times as contaminated as did surface specimens. A concentration factor of over 100,000 was estimated for DDT and PCB in soft mussel tissues.

BIOLOGICAL CONTAMINANTS

Mussels and other shellfish are efficient concentrators of bacteria and viruses from seawater. SCCWRP (1307), in testing mussels (Mytilus californianus and Mytilus edulis) from three nearshore locations, detected enteroviruses in mussels from a channel where coliforms in the mussels exceeded 275,000/g (most probable number) of digestive gland tissues. However, no enteroviruses were detected in mussels from a beach where coliforms in the mussels were about 30 times lower, or 8,600/g (most probable number). The beach site was at the base of a large sewage outfall system, yet microbial contamination was quite low compared to that of samples from the channel, which was much further removed from a known source of contamination.

Further experiments involving the suspension of caged mussels in outfall areas were conducted by Morris et al. (973). Viruses were detected and quantified in 16 of 39 samples taken from mussels suspended at various depths. Table 137 lists concentrations of total coliforms and viruses in these 16 samples. It also shows that the ratio of coliforms to virus particles ranged from 11,400 to 10.1 million. Calculations indicated that viruses were more concentrated with respect to coliform bacteria in mussels than in sewage and that mussels positive for viruses contained 0.1 to 8 virus particles per individual mussel.

A survey by the EPA (1241) of shellfish beds in San Francisco Bay reported bacterial levels in shellfish from 16 sampling stations. Total coliform, fecal coliform, and salmonella densities were determined. Shellfish from the bay commonly had fecal coliform levels that exceeded the standard of 230/100/g of meat set by the National Shellfish Sanitation Program. Fourteen of 16 stations sampled were in violation on at least one occasion. A high value of 23,000 fecal coliforma/g of shellfish meat was found at two locations. Salmonella kentucky and Salmonella typhimurium, pathogenic bacteria, were each found at one location.

There have been several disease outbreaks associated with the consumption of shellfish contaminated with wastewater microorganisms. Bryan (161) in a literature review cited 10 instances in which shellfish, particularly oysters, transmitted typhoid fever to humans; 13 instances in which

TABLE 137. CONCENTRATIONS OF TOTAL COLIFORMS AND ENTEROVIRUSES IN SEAWATER AND IN DIGESTIVE GLANDS OF MYTILUS CALIFORNIANUS SUSPENDED FROM BUOYS (973)

Location and Water Depth	Date Sampled	Total Coliform Seawater (No./ℓ)	Total Coliform Mussels (No./kg)	Viruses in Mussels (PFU/kg)*	Ratio Coliforms to Viruses in Mussels
Santa Monica Bay					
15 m	6 Mar 76	4×10^3	7.9×10^6	263	3×10^4
45 m	6 Mar 76	8×10^3	3.5×10^7	140	2.5×10^5
45 m	11 Feb 76	ND**	7×10^7	30	2.33×10^6
Palos Verdes					
Buoy 1					
30 m	1 Mar 76	7×10^3	1.1×10^7	114	9.65×10^4
Buoy 2					
16 m	3 May 76	1×10^2	3.48×10^7	620	5.61×10^4
16 m	17 May 76	1.2×10^3	1.2×10^8	552	2.17×10^5
32 m	3 May 76	5.9×10^4	3.48×10^7	1,475	2.36×10^4
32 m	17 May 76	7×10^6	7.8×10^6	210	3.71×10^4
32 m	1 Jun 76	8.4×10^3	8.6×10^6	321	2.68×10^4
Buoy 3					
17 m	3 Oct 75	5×10^2	4.9×10^5	41.7	1.18×10^4
25 m	30 Oct 75	3.7×10^2	1.2×10^6	105.3	1.14×10^4
32 m	3 Oct 75	1.1×10^3	8×10^6	315.8	2.53×10^4
32 m	30 Oct 75	1.7×10^3	4×10^6	263.2	1.52×10^4
Orange County					
Surface	6 Feb 76	ND	9.2×10^8	91	1.01×10^7
15 m	6 Feb 76	ND	9.2×10^8	476	1.93×10^6
45 m	6 Feb 76	ND	3.1×10^7	25	1.24×10^6

* Values given do not reflect an estimated efficiency of recovery of 35%.

** ND, no data.

clams and oysters were responsible for viral hepatitis outbreaks; and 3 instances in which cholera was transmitted by shellfish. The bacterial or viral agents of these diseases (*Salmonella typhi*, *Vibrio cholerae*, and hepatitis virus A) can be retained by shellfish as they filter large volumes of water through their bodies. This filtering action can result in the accumulation of bacteria and virus concentrations in shellfish tissues that are greater than the concentration in the water being filtered. The potential public health consequences are obvious.

When two-year-old California mussels were placed in water containing approximately 3.5×10^4 polio virus, plaque-forming units (pfu) per mℓ, the mussels accumulated 2.6×10^4 units/g of tissue after 48 hr. West Coast shore crabs, after residing in water containing 1.4×10^4 polio-virus pfu/mℓ for 48 hr, accumulated 4.9×10^3 pfu/g of tissue. Crabs fed virus containing mussels accumulated 74 to 94 percent of the virus present in the mussels (322).

Coxsackie virus survival in live oysters was studied by Metcalf and Stiles (905). Oysters exposed to water containing virions in the laboratory were then immersed in an estuary for two months. Virions survived in the oysters for the two months during the winter, but less than one week during summer. At no time were virions isolated from control oysters. It was postulated that the shellfish were dormant in cold water during winter, favoring virus retention, but that warmer summer water temperatures brought on oyster feeding activity and depuration of virions.

Virus survival in chilled, frozen, and processed oysters has also been studied (DiGirolamo et al., 323). Polio virus was found to survive in raw oysters refrigerated at temperatures between 5 and -17.5°C for 30 to 90 days. The survival rate varied from 10 to 13 percent. The survival rate of virions in the oysters that withstood stewing, frying, baking, and steaming ranged from 7 to 10 percent. It was concluded that, if they are harvested from contaminated areas, not only fresh, but also refrigerated and cooked oysters can transmit virus diseases.

CROPS

INTRODUCTION

Disposal of wastewater and sludge to agricultural land is presently practiced at over 400 locations in the United States. The majority of these locations grow crops that are not consumed directly by man, e.g., forage, animal feed, etc. There are, however, examples of municipal waste application to over 25 different edible fruits and vegetables.

This municipal waste application can cause surface contamination of plants by viable pathogenic microorganisms, as well as accumulation of various trace metals in plant tissues. There is some concern over the possibility that humans may consume pathogenic bacteria and viruses or hazardous amounts of metals when they eat crops grown on waste-supplied land. There is also concern over the adverse effects of feeding such crops or contaminated forage to animals intended for human consumption. Identification of the contaminant levels that may be associated with the surfaces or tissues of crops grown on waste-supplied soils is still in progress. The pertinent literature reviewed is listed in Table 138.

WATER QUALITY PARAMETERS

High nitrate content in food has been identified as a potential public health concern (133, 1432); however, no research was found in the course of this review which indicates that nitrate accumulation to high levels is a problem in crops grown on sludge or wastewater-amended soil. On the contrary, nitrate uptake by crops is generally considered desirable, since it protects groundwater from nitrate contamination.

Although municipal sewage sludge contains significant quantities of nitrogen, the amount of nitrogen that is made available to plants at typical land application rates does not approach the amount that is made available by inorganic nitrogen fertilizers. Viets and Hagemen (1432) examined the potential for nitrate contamination of crops resulting from the use of inorganic nitrogen fertilizers. They concluded that the current heavy use of chemical fertilizers has caused no overall increase in the nitrate content of foods and livestock feeds.

ELEMENTAL CONTAMINANTS

Applications of sanitary wastes to soils affect the metal content of crops directly by serving as a source of trace metals

TABLE 138. LITERATURE REVIEWED PERTAINING TO CROPS

Contaminant	Reference Number
Water Quality Parameters	
Ammonia	267, 272, 352, 366, 563, 586, 665, 666, 755, 1299, 1347, 1556
BOD	272, 352, 361, 485, 916, 1215, 1556
COD	87, 190
Chlorides	55, 56, 272, 362, 365, 665, 666, 684, 1072
Fluorides	142
Iodides	1229
Nitrates	55, 56, 133, 151, 190, 267, 362, 366, 500, 579, 586, 658, 665, 666, 684, 690, 755, 916, 1000, 1042, 1072, 1115, 1124, 1141, 1256, 1299, 1347, 1381, 1424, 1432, 1440, 1556
Nitrites	55, 56, 352, 366, 690, 755, 1124, 1299, 1556
Phosphates	55, 56, 87, 116, 190, 267, 272, 352, 366, 684, 690, 711, 755, 916, 1042, 1072, 1115, 1140, 1347, 1409, 1424
Suspended solids	87, 190, 352, 361, 485, 1215, 1556
Total dissolved solids	55, 56, 272, 352, 1215, 1256
Total organic carbon	352, 877
Other (general)	55, 56, 551, 552, 665, 666, 1376
Elemental Contaminants	
Aluminum	366, 634, 710, 720, 1108
Arsenic	267, 366, 720, 790, 1040, 1042

TABLE 138(continued)

Contaminant	Reference Number
Barium	267, 1040, 1042, 1421
Beryllium	267
Boron	137, 211, 267, 272, 352, 366, 528, 581, 684, 939, 1040, 1088, 1113, 1347, 1495
Cadmium	56, 151, 190, 211, 212, 232, 267, 366, 417, 433, 451, 581, 586, 619, 690, 710, 724, 737, 854, 1040, 1105, 1124, 1144, 1381, 1424, 1556, 1557
Chromium	151, 211, 212, 267, 366, 417, 451, 586, 690, 737, 854, 1040, 1042, 1105, 1109, 1124, 1424, 1556, 1557
Cobalt	211, 267, 366, 634, 1040, 1042, 1422
Copper	40, 55, 56, 151, 190, 211, 212, 232, 267, 366, 417, 451, 454, 528, 583, 634, 690, 710, 737, 854, 916, 939, 1038, 1040, 1088, 1105, 1124, 1144, 1154, 1229, 1424, 1495, 1557
Germanium	595
Iron	212, 232, 267, 366, 417, 451, 528, 580, 583, 634, 665, 666, 710, 737, 854, 916, 939, 1038, 1105, 1229, 1409, 1424, 1444, 1495, 1557
Lead	143, 151, 190, 211, 212, 267, 366, 417, 451, 634, 710, 737, 854, 1040, 1042, 1124, 1144, 1409, 1556
Manganese	190, 212, 267, 366, 417, 451, 580, 581, 583, 634, 701, 703, 710, 744, 939, 1040, 1042, 1105, 1124, 1229, 1409, 1495, 1556
Mercury	151, 211, 212, 366, 417, 451, 1040, 1042, 1160, 1295, 1557
Molybdenum	267, 366, 648, 711, 1040, 1042, 1050, 1229

TABLE 138(continued)

Contaminant	Reference Number
Nickel	55,56, 151, 190, 211, 212, 267, 366, 417, 583, 634, 690, 710, 1040, 1042, 1124, 1144, 1423, 1424, 1556, 1557
Selenium	124, 366, 489, 518, 711, 743, 993, 1001
Tin	267, 1445
Uranium	7
Vanadium	7
Zinc	55, 56, 151, 190, 211, 212, 217, 267, 273, 351, 353, 366, 417, 451, 528, 581, 583, 586, 619, 634, 701, 703, 710, 724, 737, 854, 916, 939, 1038, 1040, 1105, 1124, 1144, 1229, 1381, 1409, 1424, 1468, 1495, 1557
Other (general)	55, 151, 205, 211, 212, 267, 365, 417, 485, 583, 586, 619, 634, 918, 1042, 1098, 1110, 1299, 1381, 1409, 1556, 1557
Biocidal Contaminants	
Aldrin	916
Chlorinated hydrocarbons	918, 1397, 1534
DDT	17, 453, 916, 1534
Dieldrin	916, 1534
Herbicides	916, 1256
Synthetic/Organic Contaminants	16, 934, 1256, 1457, 1564
Biological Contaminants	
Adeno virus	916
Bacteria	74, 161, 399, 428, 665, 666, 738, 913, 918, 1184, 1234

TABLE 138 (continued)

Contaminant	Reference Number
Coliforms	17, 155, 200, 357, 399, 459, 665, 666, 913, 916, 1006, 1184, 1234, 1556
Coxsackie virus (A&B)	161, 357
ECHO virus	357
Escherichia coli	200, 913, 1184, 1189
Fecal streptococci	200, 656, 913
Hepatitis virus	357, 459, 916, 918
Leptospirosis	200, 357, 913, 1189
Mycobacterium	200, 913, 916, 1234, 1361
Parasitic worms	161, 200, 417, 459, 913, 918, 1186, 1188, 1189, 1234
Polio virus	161, 357
Protozoa	417, 459, 913, 1184, 1185, 1189, 1234
Salmonella	161, 200, 357, 361, 459, 913, 916, 918, 1006, 1184, 1189, 1234
Shigella	161, 200, 459, 685, 913, 1006, 1184, 1189
Vibrio cholerae	161, 198, 200, 459, 916
Virus	161, 428, 459, 485, 738, 913, 983, 1184, 1256
Other (general)	46, 161, 433, 665, 666, 913, 918, 1030, 1256, 1556

and indirectly by altering the soil chemistry, which in turn alters the solubility and mobility of metals (1409).

Many metals are essential micronutrients for plant and animal growth. Increasing their levels in the soil solution may remedy deficiencies in the plant "diet," resulting in an improved food supply that can remedy trace element deficiencies in animal diets. There are, however, limits to the amounts of trace metals that plants and animals can accumulate without showing symptoms of metal toxicity. Selenium, for example, is easily accumulated by certain plants that are adapted to growth in soils which naturally contain relatively large amounts of this element. The selenium doesn't affect the plants, but is toxic to livestock eating the plants. Urinary selenium levels appear higher in humans ingesting foods raised in naturally seleniferous soils, and chronic and acute cases of poisoning have been reported (1114).

The levels of various metals that are toxic to plants vary with plant species and soil conditions and have not been fully outlined. The levels that are safe in plants entering the human food chain are also undecided in many cases. According to several investigators (151, 211, 1042), the elements that pose a potential hazard to the food chain through plant accumulation are cadmium, copper, boron, nickel, molybdenum, and zinc.

Municipal sludges usually contain trace elements in concentrations higher than those found in typical agricultural soils (586). Small applications of sludge would normally be sufficient to correct any trace element deficiencies of the soil. Land application as a form of sludge disposal, however, is practiced on a continuing basis; therefore, large cumulative quantities of sludge are applied. Large metal concentrations do eventually build up in soils that receive sludge applications continually. In fact, many of the trace metals remain near the soil surface (309).

This long-term, heavy accumulation of metals within the plant root zone of the soil has led to speculation that the continued application of sludge will eventually result in adverse crop and food chain effects, even though no effects are seen in the initial years of sludge application. However, there is strong evidence that in many soils, plant-available metal remains constant from year to year despite cumulative increase in the total metal content of the soil.

From 1968 through 1973, over 160 dry tons of sludge/ac have been applied to corn at the University of Illinois Northeast Agronomy Research Center in Elwood, Illinois (1557). Metal levels in corn grown at the Elwood site have not inceased as the cumulative total of applied sludge has increased each year; the levels have only been increased by the sludge applied in any one growing season. In particular, the cadmium concentration of the corn grown did not exceed 1.1 ppm through 1973, after reaching a level of 1.0 ppm in 1970. Experiments conducted with soybeans yielded the same conclusion: plant-available metal levels in soils are not closely related to cumulative total sludge applications.

Similarly, experiments conducted at the Hanover Park research farm by the Metropolitan Sanitary District of Greater Chicago (1557) showed that after a cumulative total sludge application of 90 tons/ac in 6 yr, there were no unusual concentrations of metal in the corn grown. There was no significant difference between the metal concentrations of corn from sludge-supplied soil and from control soil receiving no sludge.

Soybeans were grown annually for 6 yr on field plots irrigated with digested sludge at three application rates (586). Applications of digested sludge significantly increased soil levels of zinc and cadmium, which was reflected by increased concentrations of these elements in plant tissues. For a particular annual loading rate, the amounts of cadmium and zinc in plant tissues reached relatively stable levels during the 6 yr. The magnitude of annual sludge applications was a more important determinant of amounts of elements in plants than were the amounts of zinc and cadmium accumulated in the soil from previous years. Some of the zinc and cadmium accumulated in the soil as a result of sludge applications in previous years was maintained in forms available to plants. But when annual sludge applications were terminated, zinc and cadmium concentrations were significantly decreased in plant tissues. Hinesly et al. therefore concluded that if hazardous concentrations of zinc and cadmium are to occur in plants as a result of metal accumulation in soils, they will occur during the time sludge is being applied rather than later as had previously been suggested.

According to Zenz et al. (1557), there is no data available to support the assumption that the long-term enrichment of soil considerably beyond normal concentrations of heavy metals will result in correspondingly high levels of these metals in plants.

Brown (151) explored this point further. He states that plant uptake of metals from soils depends on the portion of soil metal that is plant available rather than on the total metal content of soils. It is not known how to measure plant-available metal directly. Several investigators have tried to correlate plant uptake of zinc, copper, or nickel with the quantities of these metals that are extracted from soils by various salts, acids, and organic metal-complexing agents. This approach for estimating the portion of soil metal that is available for plant uptake has not been completely successful. However, the results are much more meaningful than are estimates based on water solubility considerations. Brown cites data to show that almost all of the zinc, copper, and nickel in sludge that were applied to a Blount silt loam soil over a period of 3-yr were still extractable by 0.1 N HCl at the end of the third year. He cites another 2-yr study in which over 100 percent of the applied zinc was extractable by 0.1 N HCl at the highest application rate used.

A study cited by Chaney et al. (212) reported the metal content of sludge, soil, and corn tissues at a 35-yr-old sludge disposal farm at Dayton, Ohio. The treated soils were very high in total metals: 2,065 ppm zinc, 843 ppm copper, and 70.5 ppm cadmium. Control soils contained 158 ppm zinc, 51 ppm copper, and 2.0 ppm cadmium. The corn leaves grown on the control and treated soils contained 67 and 196 ppm zinc, 10 and 40 ppm copper, and 2.1 and 13.9 ppm cadmium, respectively; the corn from control and treated soil contained 12 and 79 ppm zinc, 8 and 12 ppm copper, and 0.8 and 0.9 ppm cadmium, respectively. This data appears very encouraging in regard to the long-term low availability to corn grain of sludge-borne cadmium. A difference between cadmium buildup in leaf tissue relative to grain was very evident.

Chaney et al. (212) recognized the need for research that would examine the plant availability of sludge-applied cadmium at sites that were in use for 10 or 25 yr, rather than for 2 or 3 yr. Therefore, they located long-term sludge use sites with a wide range of soil cadmium contents, established plots to which no new sludge would be applied, and grew a number of crops representing a range of cadmium accumulation characteristics on these plots and matched control plots. Results of this important research program are discussed below.

Table 139 shows the trace element concentrations of oat grain from several sampling points at one particular long-term disposal site. These results show the effect of past sludge applications, since no new sludge was applied once the experimental plots were established. Oat cadmium at this site reached 2.1 ppm (mg/kg dry weight of plant tissue) when grown in soil containing 8.4 ppm DTPA-extractable cadmium at pH 6.4. Control soils contained only 0.10 ppm DTPA-extractable cadmium. Oat normally contains only 0.01 ppm cadmium.

TABLE 139. TRACE ELEMENT CONCENTRATIONS OF OAT GRAIN AFTER SUSPENSION OF SLUDGE APPLICATIONS AT A LONG-TERM DISPOSAL SITE (212)

Soil pH	Oat grain				
	Zn	Cd	Pb	Cu	Ni
	ppm Dry Crop				
6.9	30.3	0.22	0.71	4.0	2.1
6.6	32.8	0.10	0.61	4.2	1.5
6.4	49.8	2.13	0.56	4.7	7.5
6.2	46.0	1.32	0.56	4.3	5.6
6.1	45.6	1.68	0.50	4.8	5.8

The zinc and cadmium concentrations of corn at another site are shown in Table 140. The sampling locations of low pH yielded corn with relatively low cadmium concentrations. Soil DTPA-extractable cadmium was found to be a poor predictor of corn cadmium. Corn normally contains about 0.03 to 0.10 ppm cadmium.

TABLE 140. ZINC AND CADMIUM CONCENTRATIONS OF CORN AFTER SUSPENSION OF SLUDGE APPLICATIONS AT A LONG-TERM SLUDGE DISPOSAL SITE (212)

Soil total			Soil DTPA		Corn grain	
Zn	Cd	pH	Zn	Cd	Zn	Cd
ppm			ppm		ppm	
147	6.7	5.3	43	2.15	34	0.42
124	4.9	5.5	37	1.67	21	0.21
238	10.8	5.2	74	2.92	26	0.31
134	5.6	5.1	37	1.68	23	0.72
156	6.9	5.5	44	2.03	24	0.75
125	5.9	5.2	39	2.08	34	0.85
-	9.1	6.1	34	2.42	23	0.50
77	3.1	5.2	22	1.18	23	0.39
90	3.9	5.1	27	1.32	21	0.62
156	5.7	7.4	33	1.67	21	0.062
75	2.3	6.9	14	0.70	17	0.048

As shown in Table 141, the soil pH was high at a site where a large amount of low-metal sludge was applied. Fescue was somewhat higher in zinc and copper on sludge-amended plots, but for cadmium and lead there was no significant difference between sludge-amended and control plot fescue.

TABLE 141. TRACE ELEMENTS IN SOILS AND TALL FESCUE AT A SITE WHERE HIGH LIME FILTER CAKE SLUDGE WAS APPLIED AT 200-400 DRY T/A (212)

	pH	Zn	Cd	Pb	Cu	Ni	Mn	Fe
				ppm dry weight				
Total metal content								
Sludged soil	7.8	128.	1.01	42.0	45.6	12.4	339	--
Control soil	5.3	28.	0.08	10.4	3.1	4.8	186	--
DTPA-extractable metal								
Sludged soil	7.8	19.3	0.28	5.5	15.8	0.44	15	174
Control soil	5.3	2.3	0.07	3.2	0.9	0.41	20	81
Grass metal content								
Sludged soil	7.8	61	0.24	3.5	7.7	1.0	116	410
Control fertilized	6.2	23	0.18	6.1	4.7	1.2	87	112
Control unfertilized	6.1	13	0.11	4.6	4.1	0.18	110	194

Table 142 shows the trace element concentrations of soybean and corn and associated soils at various sampling spots at another long-term disposal site. The very low pH of the corn field may have been partly responsible for the substantial enrichment of corn above normal cadmium levels of 0.03 to 0.10 ppm. The soybean shows little or no enrichment above normal levels of 0.2 ppm cadmium.

Table 143 shows the cadmium results for chard and soybean tissues grown at sites near three different cities. The chard cadmium results at City 4 show no effect of sludge-applied cadmium. At City 13, chard cadmium was increased at low pH, but not at high pH; at City 9, it was very high (73 ppm) at low soil pH, but dropped to 5.5 ppm at high soil pH.

Soybean leaves and grain were substantially enriched in cadmium at Cities 9 and 13, raising the soil pH lowered grain cadmium about 60 percent at City 9 and about 75 percent at City 13. Soybean grain normally contains up to 0.2 ppm cadmium.

TABLE 142. TRACE ELEMENT CONCENTRATIONS OF SOYBEANS AND CORN AT A LONG-TERM SLUDGE DISPOSAL SITE (212)

Soil pH	DTPA-extractable Zn	Cd	Cu	Ni	Soybean grain Zn	Cd	Pb	Cu	Ni
	- - - ppm, air dry soil- - -				- - - - - ppm, dry crop - - - - -				
6.4	8.8	0.22	5.3	0.8	48	0.30	0.50	12.0	1.8
5.5	13.6	0.33	8.1	1.2	67	0.19	0.84	14.3	2.4
5.9	9.6	0.23	5.2	0.6	56	0.23	0.74	12.8	2.8
6.2	5.9	0.17	4.2	0.5	51	0.23	0.60	13.9	2.0
6.3	3.7	0.11	2.3	0.4	45	0.18	0.89	11.8	1.5
6.4	3.9	0.11	3.0	0.5	44	0.37	0.57	12.7	1.4

Soil	DTPA-extractable Zn	Cd	Cu	Ni	Corn grain Zn	Cd	Pb	Cu	Ni
	- - - ppm, air dry soil- - -				- - - - - ppm, dry crop - - - - -				
5.2	7.1	0.13	3.8	1.2	19	0.23	0.87	1.8	0.7
5.0	6.4	0.17	4.1	0.9	18	0.56	0.78	2.0	0.4
5.0	10.2	0.22	7.1	0.8	16	0.43	0.77	1.7	0.4

High cadmium and cadmium to zinc ratio sludges were used at Cities 9 and 13. At City 9, sludge had not been applied for nearly 3 yr at harvest time, yet soybean grain contained 3.7 ppm cadmium at low pH and 1.5 ppm at high pH. Chaney et al. came to the following conclusions regarding cadmium:

- Soil pH significantly affected crop concentrations of cadmium;
- Crops differed widely in cadmium uptake, in the relative transport of cadmium to the grain, and in the influence of pH on cadmium uptake;
- Cadmium remained available to crops, even though sludge application had ceased; and
- High cadmium and cadmium/zinc sludges produced higher crop cadmium levels than did ordinary municipal sludges.

TABLE 143. CADMIUM CONCENTRATIONS (PPM DRY CROP) OF CROP TISSUES GROWN AT THREE LONG-TERM SLUDGE DISPOSAL SITES (212)

City	Soil		Chard leaf	Soybean	
	DTPA-extractable Cd	pH		Leaf	Grain
4 Control	0.13	5.7	0.6	0.27	0.17
4 Control	0.14	6.7	0.5	0.26	0.15
4 Sludged	0.53	5.2	1.9	--	--
4 Sludged	0.57	6.2	0.6	--	--
9 Control	0.13	5.3	3.6	1.04	0.36
9 Control	0.10	6.7	1.2	0.55	0.28
9 Sludged	1.13	4.8	73.0	10.7	3.70
9 Sludged	1.19	6.6	5.5	1.87	1.51
13 Control	0.09	5.3	0.9	0.24	0.16
13 Control	0.10	6.4	0.5	0.17	0.13
13 Sludged	7.15	5.6	7.0	5.70	2.64
13 Sludged	5.45	6.6	1.8	2.38	0.65

Regarding trace metals other than cadmium, Chaney et al. concluded the following:

- Crops were not influenced by additions of sludge lead.
- With the exception of chard, crops were not higher in copper from sludge use, even though a large proportion of added copper remained DTPA-extractable.
- Where sludges high in nickel were used, plant levels of nickel were increased only at low soil pH.
- Zinc remained available over many years of sludge use and for many years after sludge use stopped. Crops differed widely in zinc uptake from the same sludge treated soil; grain zinc increased less than foliar zinc.

The results of a study cited by Page (1040) are presented in Table 144. The trace element concentrations of leeks, globe beets, potatoes, and carrots grown on soil treated with 66 t of sludge/ha/yr over a period of 19 yr are reported. The carrots were grown on soil that had not received sludge for 7 yr, yet metal concentrations in carrot tops were higher than the concentrations in carrot tops from untreated soil.

TABLE 144. TRACE ELEMENT COMPOSITION OF CROPS GROWN ON SOIL TREATED WITH SLUDGE AT AN AVERAGE RATE OF 66 METRIC TONS/HA/YR FOR 19 YRS* (1040)

	Concentrations of Trace Element in Plants				
Treatment	Chromium	Copper	Molybdenum	Nickel	Zinc
	----------------µg/g dry matter----------------				
			Leeks		
Untreated	0.71	5.7	0.50	2.0	46
Treated	0.54	16.0	1.10	7.0	135
			Globe Beet Tops		
Untreated	0.85	8.5	0.45	3.2	169
Treated	1.0	10.0	0.65	16.5	> 505
			Globe Beet Roots		
Untreated	0.3	11	0.1	1.65	102
Treated	0.8	18	0.25	13.0	250
			Potato Tops		
Untreated	1.70	4.2	0.37	1.7	90
Treated	3.0	8.2	1.0	5.2	120
			Potato Roots		
Untreated	0.09	9.5	0.40	0.25	30
Treated	0.03	9.5	0.27	0.57	27
			Carrot Tops		
Untreated	0.41	8.2	0.58	1.28	47
Treated	0.88	9.9	0.84	3.0	99
			Carrot Roots		
Untreated	0.03	6.3	0.12	-	34
Treated	0.06	4.6	0.12	-	42

* Leeks, beets, and potatoes are means of two plots. Carrots were grown on soil 7 yr after treatments were discontinued.

In contrast to the studies of long-term disposal sites by Chaney et al. (212), Zenz et al. (1557), and Hinesley et al. (586), several researchers have presented data from short-term sites, where sludge was applied during the same year in which crops were grown. This type of data cannot show long-term effects.

In 1973 and 1974 corn plants were grown on digested sludge-amended soils and analyzed for their trace metal concentrations (365). In both years, sludge application increased the cadmium concentration in the whole corn plant but had no effect on the grain concentration.

Baier (55) conducted a study that involved one-time sludge applications at three rates to irrigated dryland pasture plots. Results are presented in Table 145.

TABLE 145. AVERAGE CONCENTRATIONS OF TRACE METALS IN FORAGE FROM DRY AND IRRIGATED PASTURE PLOTS RECEIVING SLUDGE AT THREE APPLICATION RATES (55)

AVERAGE CONCENTRATIONS IN DRYLAND FORAGE

(mg/kg dry wt of plant tissue)

Application Rate	22 T/A	12 T/A	3 T/A	0 T/A
Cadmium	.15	.40	.19	.12
Copper	7.61	11.03	7.67	6.55
Nickel	3.26	4.96	1.64	2.73
Lead	.66	1.38	1.21	.66
Zinc	51.63	74.42	39.43	30.02

IRRIGATED LAND FORAGE

Application Rate	22 T/A	12 T/A	3 T/A	0 T/A
Cadmium	.24	.22	.21	.09
Copper	3.39	7.34	4.51	7.22
Nickel	5.08	5.65	3.38	2.78
Lead	.80	.91	.85	.78
Zinc	53.67	58.76	41.10	27.20

The data show that only cadmium and zinc were found in consistently higher concentrations in forage from sludge-treated rather than control plots; however, the concentrations did not increase proportionately to the amount of sludge applied, suggesting that the metals were being complexed in an unavailable form.

Cadmium was held in a very shallow layer of soil, probably the top ½ in. The resulting high concentration may have been toxic enough to plant roots so that the roots did not grow in that zone, making the metal less available to the plant.

Solid municipal waste was applied to Sagehill sand at rates of 0, 100, 200, and 400 tons/ac (267). Sewage sludge was applied at 55 gal/ton of solid waste. Winter wheat was planted as a cover crop in October 1971 and was followed by spring-seeded fescue and alfalfa. Ammonium sulfate fertilizer was applied, and the plots were irrigated for two crop seasons. Wheat harvested in June 1972, and all fescue and alfalfa contained 100 to 300 ppm iron. Some alfalfa samples, especially in September 1972, contained up to 900 ppm iron. These samples raised the averages and may indicate local spots of high iron concentration in the soil. However, alfalfa may normally contain up to 1,000 ppm iron. Iron uptake in general did not increase with the addition of municipal wastes.

The manganese content of the fescue and alfalfa harvested in June increased with waste application and with nitrogen fertilization. Fescue from plots that received the highest cumulative nitrogen contained 300 to 400 ppm manganese, approaching levels toxic to the plants. Manganese availability increased with decreasing pH.

The copper content of the wheat, fescue, and alfalfa ranged between 5 and 15 ppm. Normal plant copper levels range between 5 and 20 ppm.

The zinc content of the fescue and alfalfa also increased wtih waste application and nitrogen fertilization. The fertilizer increased zinc uptake primarily by its effect on soil pH. The zinc content of both crops grown on plots that received 400 tons solid waste/ac was over 100 ppm. Zinc levels in plants grown on control plots were closer to 25 ppm.

Plant content of iron and copper was not affected by the waste treatments. Manganese and zinc uptake by wheat, fescue, and alfalfa increased with waste addition and with nitrogen fertilization. The copper content of the wheat, fescue, and alfalfa ranged between 5 and 15 ppm. Normal plant copper levels range between 5 and 20 ppm. Apparently copper added in the waste materials is released very slowly and/or is rapidly immobilized in the soil, remaining unavailable to plants.

The boron content of wheat sampled in April 1972 increased dramatically with addition of municipal wastes.

Wheat from control plots contained 4 to 6 ppm boron while wheat from plots that received 100 tons solid waste/ac contained a moderately phytotoxic level of about 20 ppm boron. Wheat from plots that received 200 or 400 tons solid waste/ac contained up to 100 ppm boron. The normal boron content of wheat is 3 to 5 ppm. Boron uptake by alfalfa increased only slightly.

Molybdenum uptake by alfalfa grown with the higher waste treatments reached levels potentially hazardous to livestock during the first growing season but decreased to safe levels (2.0 ppm) the second year. The highest level during the first season was 10.8 ppm molybdenum. Molybdenum is toxic to livestock at 5 ppm.

The cobalt content of wheat, fescue, and alfalfa ranged from 0.4 to 0.7 ppm. Normal levels are 0.02 and 0.29 ppm cobalt in alfalfa, fescue, and wheat.

Chromium uptake by fescue and alfalfa was affected very little by waste additions.

Digested sludge has been applied to a land reclamation site at Fulton County, Illinois, since 1971 (1556). Crop tissue samples have been taken each year since the project began. The metal content of corn grown on nonsludged and on sludged fields was compared in 1972 and 1973. The amount of sludge applied ranged from 0.49 to 5.8 dry tons/ac. The data from the analyses reveals no significant difference between the sludged and nonsludged fields in the levels of zinc, iron, manganese, copper, nickel, chromium, lead, and cadmium in corn grain.

A 1-yr study of five existing sludge disposal sites receiving various quantities of sludge was conducted by Keeney et al. (690). Cadmium, chromium, copper, nickel, and zinc were measured in grass samples and in soil. The concentrations of these metals were increased in the surface soil of each sludge disposal area. The two sites receiving the highest sludge application rates -- 500 and 290 dry t/ha of sludge -- were of lower pH (4.6 and 4.4). At these two sites, trace element uptake by plants was greater than in the control areas. At other sites receiving less sludge, the trace element concentrations of grasses, except for zinc, compared more favorably to grasses in the control areas. At two of these sites, the average zinc concentration of grass increased when compared to the control areas, from 28 to 80 ppm and from 16 to 30 ppm.

The average cadmium, copper, manganese, and zinc content of the grass collected from the site receiving the most sludge increased when compared to the control samples, from 0.2 to

1 ppm, 5.8 to 14 ppm, 70 to 240 ppm, and 19 to 290 ppm, respectively. The average chromium and nickel content of grass compared closely with the control samples.

Dowty et al. (1409) studied the uptake of metals by seven vegetable crops after 0, 112, 225, and 450 tons/ha of sludge was applied to coarse sandy soil. Generally, metal contents of the vegetable tissues were higher than those of fruiting, root, and tuber tissue. In most edible tissue, heavy metal accumulations did not increase more than two- to threefold as a result of amending the soil with 450 tons/ha sludge. Lettuce tissue was an exception, with increases of 11-, 7.5-, and 4.4-fold for zinc, copper, and cadmium, respectively. Lettuce seems to be an accumulator of metals, whereas potatoes and carrots seem to be excellent nonaccumulators. Approximately three times as much zinc accumulated in pea vine tissue as in edible fruit. The cadmium to zinc ratios of edible tissues of potatoes, carrots, and peas were less than 0.01 at all sludge application rates. The ratio for lettuce was 0.029 for the control treatment of 0 tons of sludge applied, and 0.012 for the highest application rate.

Shredded municipal refuse and primary sludge were applied to Sagehill loamy sand before the planting of wheat, fescue, and alfalfa during a 1-yr study conducted by Halvorson (528). The concentrations of iron, copper, boron, and zinc in fescue, and of iron, molybdenum, boron, and zinc in alfalfa were affected by the waste treatments. Results of the study are shown in Table 146.

TABLE 146. AVERAGE TRACE ELEMENT CONCENTRATIONS (mg/kg DRY WT PLANT TISSUE) IN FESCUE AND ALFALFA GROWN ON PLOTS RECEIVING 400 T/A SHREDDED MUNICIPAL WASTE PLUS SLUDGE AND ON CONTROL PLOTS (528)

Sludge Application Rate	Fescue				Alfalfa			
	Fe	Cu	B	Zn	Fe	Mo	B	Zn
0 gal/A	122	9	3.4	29	190	0.9	34	44
200 gal/A	208	25	37	268	452	14.5	60	320

Table 147 shows the heavy metal concentrations in the seeds of soybeans that received a maximum of 228 mm of liquid digested sludge during the year 1970. Sludge was applied to plots at 0, 25, 50 or 100 percent of the maximum loading rate (1557).

TABLE 147. AVERAGE TRACE METAL CONCENTRATIONS OF SOYBEAN SEEDS FERTILIZED WITH DIGESTED SLUDGE AT VARIOUS APPLICATION RATES (1557)

Percent of Maximum Application Rate	Metal Concentration (mg/kg dry wt) Zinc	Copper	Nickel	Cadmium
0%	59	13.1	6.1	0.12
25%	68	14.9	9.4	0.38
50%	73	12.5	10.5	0.60
100%	83	12.2	10.5	1.08

An examination of this data shows that zinc, nickel, and cadmium contents of soybean seeds were much increased by sludge application. Copper contents in plant tissues were not increased above the levels in plants from control plots.

Sludge application to pasture from August 1, 1973 through September 1974 was studied by Fitzgerald and Jolley (417). Sudan pasture grass, corn stubble, rye pasturage, corn silage, and alfalfa hay were analyzed for chromium, copper, iron, lead, nickel, manganese, and zinc. Increases of metal concentrations in plants other than the Sudan grass were not apparent over the test period. The metal concentrations found in Sudan grass are reported in Table 148. In most cases, control Sudan grass contained higher peak levels of metals than test Sudan grass.

TABLE 148. TRACE ELEMENT CONCENTRATIONS IN SUDAN GRASS GROWN ON SLUDGE-IRRIGATED AND CONTROL PLOTS (mg/kg OVEN-DRIED SUDAN) (417)

Trace Metals	Sludge-Irrigated	Control
Zinc	42.4 - 98.1	69.7 - 100
Copper	4.63- 33.0	1.0 - 20.0
Chromium	5.37- 39.6	1.95- 20.0
Iron	177 -2,666	484 -11,794
Nickel	2.98- 11.2	2.68- 20.3
Lead	3.81- 13.4	3.40- 19.3
Manganese	32.8 - 96.2	77.6 - 251.0

Chaney et al. (212) suggested that farmers must be educated to use sludge properly. Presently, farmers are often not aware that sludge has a tendency to lower soil pH and that liming the soil can remedy the situation. This is important because increased plant uptake of metals has been observed at low pH. Nonuniform spreading practices used on fields by farmers can result in localized regions of extremely high metal content, especially because added metals tend to remain in the surface layers of the soil.

BIOLOGICAL CONTAMINANTS

In Europe and other parts of the world outside the United States, epidemics of typhoid and infectious hepatitis have reportedly been caused by the application of human wastes to agricultural crops (361). These outbreaks have been attributed to the consumption of raw vegetables fertilized by night soil (human feces) or grown on farms irrigated with raw wastewater. Night soil used on vegetables and fruits has been responsible for recurrent outbreaks of amoebic dysentery, bacillary dysentery, gastroenteritis, cholera, and roundworm (ascaris) in the Orient (459). Sewage irrigation of vegetable crops was found to be linked with an outbreak of cholera in Jerusalem in 1970 (198).

A partial summary of the disease outbreaks linked to foods contaminated with sewage is presented in Table 149, adapted from Bryan (161). These outbreaks results from the surface contamination of vegetables with enteric pathogens. Normal, healthy vegetables are internally sterile, even when their surfaces and the surrounding soil are contaminated with bacteria and viruses (1184).

Bryan states that outbreaks of food-borne illness may continue to occur sporadically, if raw or partially treated wastewater is used for irrigation of foods that are consumed raw (161). The true extent of disease infection from raw produce is unknown (459). Many infections may be experienced as mild intestinal upsets that do not require medical treatment and therefore go unreported. In the United States, the most recently reported disease outbreak associated with wastewater irrigation occurred in 1919. Allegedly, blackberries or vegetables grown on a sewage farm at Pasadena, California, caused typhoid in eight employees (200).

Enteric pathogens survive some stages and sometimes the entire process of wastewater treatment. Viable pathogens are therefore applied to the land in the process of irrigation with wastewater. After application, these pathogens must survive long enough to be present on the crops when they are harvested. Adverse environmental conditions considerably reduce the survival rate of these organisms. Salmonella, shigella, enteropathogenic *Escherichia coli*, amoebic cysts, ascaris ova, and enteroviruses have been detected on garden produce growing in soil contaminated or irrigated with sewage effluents, but

TABLE 149. OUTBREAKS ASSOCIATED WITH FOODS CONTAMINATED BY SEWAGE OR WASTEWATER (Adapted from 161)

Disease	Source of Contamination	Food
Typhoid fever	Sewage sludge fertilizer and irrigation	Celery
Typhoid fever	Sewage-contaminated watercress beds	Watercress
Typhoid fever	Human manure	Rhubarb
Typhoid fever	Privy-polluted watercress beds	Watercress
Typhoid fever	Sewage irrigation	Vegetables, blackberries
Typhoid fever	Sewage irrigation	Raw vegetables
Amebiasis	Sewage irrigation	Vegetables
Taeniasis	Sewage irrigation	Beef
Typhoid fever, Paratyphoid fever	Secondary sewage treatment (activated sludge)* plant effluent irrigation and wash water	Vegetables
Shigellosis	Irrigation water (Primary treatment plant effluent)**	Cabbage
Amebiasis	Night soil	Vegetables
Fascioliasis	Sewage-polluted water+	Watercress
Taeniasis	Human feces contaminated trench silo	Rare beef
Fascioliasis	Animal feces contaminated watercress bed	Watercress
Salmonellosis (Animal infection)	Animal dung slurry used to irrigate pastures	Grass

TABLE 149 (continued)

Disease	Source of Contamination	Food
Fascioliasis	Animal feces contaminated water used for watercress beds	Watercress
Ouch-ouch disease (cadmium poisoning)	Mining waste used to flood rice fields	Rice
Fascioliasis	Animal feces contaminated watercress bed	Watercress
Viral hepatitis	Septic tank effluent contaminated water**	Watercress
Salmonellosis (Animal infection,* possible human cases)	Human sewage flowing over grazing land	Grass
Viral hepatitis	Sewage-polluted water+	Oysters
Cholera	Raw sewage irrigation	Vegetables

* Secondary treated sewage
** Primary treated sewage
+ Implied

their detection is infrequent (459). When pathogens are found, they are found in a low percentage of samples and usually for only a short time after irrigation (161).

Studies have been made on the viability of various indicator and pathogenic organisms on crops irrigated with wastewater. The viability of such organisms varies from several days to a few months, depending on the type of organism and its resistance to environmental factors e.g., climatic conditions, soil moisture, amount of protection provided by crops, etc. Research has shown that salmonella may persist for up to 70 days in soil irrigated with sewage under moist winter conditions, and for about half that period under drier summer conditions (1256).

In 1949, Falk (399) conducted experiments at a small sewage irrigation farm outside a southern New Jersey state institution. Three plots were established for the growth of tomatoes. One plot was treated throughout the growing season with raw, settled sewage; the second was treated with sewage only before planting; and the third was never treated with sewage. Similar coliform densities were found on the surfaces of normal tomatoes from all three plots, indicating no significant contamination from the sewage. When the stem ends of the tomatoes were abnormally split, however, high coliform densities were found on tomato surfaces. On broken tomatoes, the coliform density was three times higher when the soil was treated with sewage during growth than when it was treated prior to planting or not at all.

In later experiments at this site, Rudolfs et al. (1184) applied feces or Escherichia coli suspensions directly to the surfaces of growing tomatoes. By the end of 35 days, the residual coliform concentration on the surfaces decreased to or below that of uncontaminated controls. The representatives of salmonella and shigella on tomato surfaces in the field did not survive more than 7 days, even when the organisms were applied with fecal organic material, which might have afforded some protection. The investigators concluded that cessation of sewage application to crops one month before harvest will offer a considerable margin of safety for protection against enteric bacterial diseases even though vegetables are consumed raw.

Rudolfs et al. (1189) found that the maximum survival time of Endamoeba histolytica on lettuce or tomatoes in the field was 18 hr. Ascaris eggs, in some cases, remained on tomatoes for more than a month after application, but were unable to develop into a mature, infective state. It appeared, therefore, that cessation of sludge application to crops a month before harvest should greatly reduce the probability of transmission of ascaris, one of the hardier parasitic worms.

According to Foster and Engelbrecht (428), cessation of sludge application to crops a month before harvesting probably would not offer safety against the transmission of Mycobacterium Tuberculosum. They reported that when wastewater containing tubercle bacilli was applied to a radish patch, some bacilli were discovered after three months of exposure to the elements.

The use of settled chlorinated sewage effluent for vegetable crop irrigation was studied. Over 50 percent of the irrigation water samples tested contained salmonella bacteria, ascaris eggs, or Endamoeba coli cysts. Yet only 1 of the 97 samples of vegetables tested yielded salmonella, and ascaris eggs were recovered from 2 of 34 vegetable samples tested. Ascaris was found only when raw sewage was mixed into the irrigation water.

Well-digested sludge may contain fewer viable pathogenic organisms than raw sludge and therefore present a lesser hazard when land applied. Carroll et al. (200) cited a study of the persistence of pathogens in liquid-digested, sludge-amended soil. Fecal coliforms were found in the soil 18 weeks after application of liquid-digested sludge; Pseudomonas aeruginosa were found after 16 weeks, and salmonella were found for up to 19 weeks after application. Before application, the soil had contained none of these bacteria.

On the basis of his literature review, Shuval (1256) concluded that sufficient numbers of pathogens can survive under the conditions normally expected in agricultural practice to result in a potential health hazard, if crops recently irrigated with wastewater are consumed without cooking. However, he states that no clear-cut epidemiological evidence is available to indicate that the carefully regulated use of wastewater to irrigate crops not used for direct human consumption or crops consumed only after cooking and processing has ever led to disease outbreak. In addition, no records exist of diseases having been caused by using digested sludge as a soil conditioner or fertilizer (339).

Irrigation practices play an important role in minimizing the health risks involved in the use of wastewater in agriculture. For instance, surface irrigation techniques applied to fruit trees can produce uncontaminated fruit presenting no public health risk. However, if spray irrigation is used, there is a definite possibility of contaminating the fruit. Similarly, certain vegetable crops might be irrigated by the ridge and furrow technique in such a way as to avoid any direct contact between the wastewater and the crop.

Food-borne disease organisms must survive processing and cooking steps,and often have to multiply to reach infective levels. If wastewater-contaminated foods are cooked so that the contaminated portions reach 165°F, vegetative bacteria (but not spores), protozoa, helminth eggs, and most viruses will be killed. In South Africa, only three cases of enteric fever were observed over a period of 9 yr in 28,000 mine workers consuming cooked, sewage-irrigated vegetables. The vegetables were cooked in a thick stew for about 20 min.

Little work has been done on the transmission of enteric viral diseases by foods contaminated with wastewater (1256). Viruses as a group, however, are generally more resistant to environmental stresses than many of the bacteria (428), and therefore there is a potential for the spread of waterborne viral diseases, including infectious hepatitis or poliomyelitis, by this route. Once again, however, proper irrigation practices and the use of wastewater only for raising foods that are not consumed raw can reduce the hazard.

LIVESTOCK

INTRODUCTION

Pasture, forage, and grain for livestock consumption are often grown with wastewater or sludge applied either to soil or crop. However, there is little information on the possible accumulation of contaminants in the livestock consuming these feeds (200). The levels to which contaminants accumulate before the animals show symptoms of toxicity are not of direct public health interest. But before contaminant concentrations reach such levels, they may reach levels at which animal products can become unsafe for incorporation into the human diet and may, therefore, be of public health concern. What these levels are, and what the likelihood is that contaminants will accumulate to these levels in livestock are basically unexplored questions. The pertinent literature that has been found is recorded in Table 150.

ELEMENTAL CONTAMINANTS

Trace metals in small quantities are required for metabolic activities of animals. Copper and zinc are frequently added to pig and poultry feeds; livestock eating forage on selenium-deficient soils have been known to have severe deficiency diseases (1114). Therefore, in certain instances, it can be advantageous to add trace metal-bearing municipal waste to livestock pasturage.

Although the ingestion of feed containing excessively high concentrations of trace metals is known to produce diseases in livestock, no adverse effects from land application operations involving municipal wastes have been reported in the literature. Cottrell (267) found that when 200 tons of solid waste/ac and 11,000 gal of municipal sludge/ac were applied to Sagehill sand, alfalfa raised thereon contained 5 to 6 ppm molybdenum. These elevated levels of molybdenum are possibly high enough to induce a copper deficiency disease in cattle.

Spray application of fluid sludges to pastures and hay fields is an important component of most all-season, all-weather fluid sludge systems. Adherence of sludge particles from spray application onto grass surfaces can increase levels of persistent organic compounds far above those reached when sludge is not sprayed directly onto pasturage. A study

TABLE 150. LITERATURE REVIEWED PERTAINING TO LIVESTOCK

Contaminant	Reference Number
Elemental Contaminants	
Boron	1113
Cadmium	737
Chromium	737
Copper	737
Lead	737
Mercury	1295
Molybdenum	267, 1040
Selenium	1114
Zinc	737
Other (general)	212, 918
Biocidal Contaminants	
Chlorinated hydrocarbons	918, 998
SS synthetics	417, 1564
Biological Contaminants	
Hepatitis	918
Salmonella	161, 200, 913, 918
Shigella	161, 200, 913
Lepto	200
Escherichia coli	147, 200, 913
Vibrio cholerae	200
Mycobacterium	288, 913

TABLE 150 (continued)

Contaminant	Reference Number
Fecal streptococcus	200, 913, 918, 1356
Protozoa	913
Parasitic worms	161, 200, 913, 918
Virus	161, 908, 913, 918
Coliforms	147, 200, 913
Bacteria	161, 913, 918, 1356
Other (general)	161

of the adherence of sludge particles directly onto grass and hay was carried out by Chaney et al. (212). They found that sludge comprised as much as 32 percent, by weight, of the harvest from the tested pasture, and rainfall was ineffective in removing the adhering sludge. Because such surface contamination can increase trace element contamination of forage, sludge use on pastures may require special technologies, such as injection into sod.

The effect of sludge application to pasture for 14 months was reported by Fitzgerald and Jolley (417). Levels of PCB's in sludge-irrigated Sudan pasture grass were measured. Sludge-irrigated pasture produced grass containing 0.1 to 1.2 µg PCB/gm = µg PCB/gm of dry plant tissue, while the grass grown on control soil receiving no sludge contained 0.2 to 0.4 PCB = µg PCB/gm. The investigators stated that little data exists with which to compare these PCB levels.

BIOLOGICAL CONTAMINANTS

The most prevalent mode of bacterial, viral, or parasitic transmission from land-applied wastes to humans appears to be through consumption of sewage-contaminated fruits and vegetables. However, certain parasites may reach humans via a parasitic life cycle involving humans, feces, and uncooked beef or pork. Livestock that have grazed on land-application sites or fed recently contaminated forage crops have high infection rates of salmonellosis, tuberculosis, ascaris, hookworm, and liver fluke.

In a review of literature, reports of beef tapeworm (*Taenia saginata*) infections of beef cattle in Great Britain fed on grasslands fertilized with sludge that had undergone mesophilic digestion for 20 days were cited (913). The mode of transmission from pasture to beef was not elaborated on, although it is known that the life cycle of the beef tapeworm involves transmission from man to feces, to cattle, and back to man. The pork tapeworm (*Taenia soluim*) has a similar life cycle involving uncooked pork rather than beef (161).

Echinococcus granulosis is another parasitic helminth found in domestic sewage. It is cyclically transmitted between sheep and dogs, but can incidentally infect humans working in sludge-supplied pastures (913).

PUBLIC HEALTH CONSIDERATIONS

EPIDEMIOLOGICAL AND PATHOLOGICAL EVALUATION OF WASTEWATER CONTAMINANTS

Many contaminants contained in municipal wastewater are known to produce adverse human health effects. A number of elemental, biocidal, and synthetic/organic constituents have been clearly identified as potential carcinogens; some have even been identified as being mutagenic or teratogenic. Although the epidemiological effects of various viruses and bacteria are known and predictable, knowledge of the health risks of certain other contaminants is incomplete or lacking. Moreover, some chemical substances work not only in isolation within the human body, but may react synergistically (two or more chemicals combining to produce a net effect that is greater or lesser than that produced if the chemicals act independently). A multiplicity of factors is involved in such reactions, and knowledge of potential health risks is scant.

Literature detailing the epidemiological and pathological effects of wastewater constituents has been surveyed; the principal health problems posed by these contaminants are summarized below. A sound understanding of human physiological reactions and consequent public health threats will better prepare responsible authorities to assess treatment performance and set acceptable standards.

Present standards controlling wastewater discharges and drinking water supplies have been assumed by many to guarantee adequate protection of public health. However, recent investigation has shown that some residual organics, carcinogenic chlorinated hydrocarbons, synthetic compounds, trace elements, and biocides are harmful even in extremely small concentrations. Existing standards are called into question by this increasing knowledge: there may be no "safe" threshold for some of these chemicals. Growing epidemiological and pathological evidence must be taken into account if discharge and drinking water standards are to safely ensure public well-being.

WATER QUALITY PARAMETERS

Suspended solids, BOD, TOC, and most other constituents of general water quality have no direct effect on public health. However, certain constituents may become associated with other, more directly harmful, contaminants. The survival and potential health threat of such contaminants may be magnified as a result of this process. Moreover, nitrogen species present in wastewater can directly affect human health.

Nitrogen

Nitrogen in wastewater effluent is usually found in one of the stable forms (ammonia, nitrate, or organic nitrogen) rather than the more hazardous nitrite form. Nitrates and nitrites occur in drugs, food, and water. Man is continually exposed to small amounts of these substances, which usually cause no harm. In high concentrations and under special circumstances, however, they may cause illness and even death. Sepp (1233) and the Hazardous Waste Advisory Committee of the EPA (1002) consider nitrite, in particular, a significant health problem.

Nitrite toxicity is the major health problem associated with these nitrogen species since nitrate easily reduces to the toxic nitrite form. Such a conversion may occur outside the human body in food or water containing nitrates. Conversion can also take place inside the body through the action of intestinal bacteria on ingested nitrates.

Nitrate/nitrite conversion that occurs during digestion requires special conditions likely to be present only in infants. The foremost prerequisite is the presence of nitrate-reducing bacteria in the upper gastrointestinal tract. Such bacteria are not normally present there, but lower in the intestinal tract. They may occur in the upper GI tract of infants, particulary those with gastrointestinal infections and a gastric pH insufficiently acidic to kill the bacteria.

Acute nitrite toxicity (methemoglobinemia) occurs when hemoglobin (the oxygen-carrying red pigment of blood) is oxidized into methemoglobin (a brown pigment incapable of carrying oxygen). Methemoglobin constitutes about 1 percent of the total hemoglobin of a healthy adult and up to approximately 4 percent of that of a healthy newborn infant. Cyanosis results when roughly 15 percent of the hemoglobin in blood is converted into methemoglobin; when methemoglobin constitutes 70 percent or more of the total hemoglobin, oxygen transport is severely impeded, and death may occur (1002).

Infants, then, are particularly prone to nitrate-induced methemoglobinemia. In addition to the presence of nitrite-reducing bacteria in the upper gastrointestinal tract as mentioned above, (1) the hemoglobin of very young infants is oxidized twice as rapidly as that of adults by nitrate to form methemoglobin; (2) the red blood cells of infants are not able to reduce methemoglobin into hemoglobin as well as adult cells; and (3) the total fluid intake of infants per unit of body weight is much greater than that of adults. Thus, for a given concentration in fluids, infants consume proportionately more nitrate than adults.

The consumption of water containing high levels of nitrates has accounted for many more cases of methemoglobinemia than all other causes combined. Methemoglobinemia of such etiology has been reported only in infants, although one study documents one occurrence resulting from the use of nitrate-contaminated well water for peritoneal dialysis in an adult. In the United States only one case has been associated with water from a public water supply; all the rest (about 300) have been due to well water (1002).

Standards for nitrates in drinking water limiting nitrate to 10 mg/ℓ expressed as nitrate nitrogen (45 mg/ℓ expressed as nitrate) were set by the U.S. Public Health Service in 1962. The 10-mg/ℓ nitrate nitrogen level was set because there had been no reports in this country of infantile methemoglobinemia associated with the ingestion of water containing nitrate at levels below 10 mg/ℓ. In addition, this level was set because it was a standard that could be met easily by most municipal water supplies. After publication of these standards, however, data reported from other countries revealed that a small percentage of cases had occurred where the water nitrate nitrogen content had been below 10 mg/ℓ.

As a consequence, the 1962 standards are currently under reevaluation. Several studies have been designed to determine more specifically the exact nitrate levels in water required to cause elevated levels of methemoglobin and clinical evidence of methemoglobinemia in infants. Preliminary results suggest that the 1962 standards provide adequate protection against clinical methemoglobinemia. However, subclinical elevations of methemoglobin have been found in infants with diarrhea or respiratory disease, consuming water with a nitrate content below the 10 mg/ℓ level (1002).

In contrast to the relative wealth of data on acute toxicity in humans, reliable data are lacking on the physiologic effects, if any, of chronic nitrate/nitrite toxicity or of mild, noncyanotic methemoglobinemia. Studies in animals indicate that nitrates and nitrites may, on occasion, cause vitamin A deficiency, and that nitrates may have an antithyroid effect by increasing the need for iodine. No data are available to indicate whether such effects can occur in man.

Chronically elevated levels of methemoglobinemia may have some effect on the human brain: abnormal changes on electroencephalograms have been observed in rats given 100 to 2,000 mg/ℓ of sodium nitrite each day for two weeks. A Russian study purports to show decreased response to visual and auditory stimuli in school children with a mean methemoglobin level of 5.3 percent of total hemoglobin. However, the study was poorly controlled, and the results are inconclusive. There have been patients with hereditary methemoglobinemia and mental retardation, but the association may be coincidental. Most patients with hereditary methemoglobinemia show no mental or neurologic abnormalities (1002).

Nitrosamines formed by the reaction between nitrites and amines have been proven hazardous to human health. Nitrites and/or precursor nitrates are found in foods, water, and drugs; amines are found in tobacco smoke, beer, tea, wine, and toothpaste as well.

Nitrosamines have potent biological effects, including acute cellular injury (primarily involving the liver), carcinogenesis, mutagenesis, and teratogenesis. To date, approximately one hundred nitrosamines have been tested in animals. The vast majority has proved to be carcinogenic. Many organs (liver, esophagus, and kidneys) that are common to diverse species of animals are susceptible to the cancer-producing effects of these compounds. These effects can be elicited experimentally by various routes of nitrosamine administration (oral, intravenous, inhalation) at extremely low doses. In some instances, cancer can develop after a single exposure (1002).

Concerns about potential nitrosamine hazards to human health arise from the possibility for (1) contact with preformed carcinogenic nitrosamines and (2) the formation of carcinogenic nitrosamines within the human body after exposure to precursor nitrites and amines. The possible formation of carcinogenic nitrosamines in the human gut through the combination of ingested nitrites and amines is of critical concern. Such reactions have been demonstrated to occur both in vitro and in vivo (in animals).

Studies in humans that were fed nitrate and a noncarcinogenic nitrosamine precursor amine (diphenylamine) have shown that diphenylnitrosamine can be formed in the human stomach. Nitrosamine determinations in these studies were made by thin-layer chromatography, a method now known to give false positive results. Unfortunately, these data have not yet been confirmed using the newer and more reliable techniques of gas-liquid chromatography and mass spectrophotometry (1002).

ELEMENTAL CONTAMINANTS

It is often difficult to assess the health effects of metals and their compounds: many metals are essential to life at low concentrations but toxic when concentrations exceed tolerance in man. The situation is further complicated because the various chemical states of metals (pure metal, inorganic or organic-metallic compounds) react differently within the body. Individual differences between subjects, incubation periods, and sites of accumulation of the substances in the body are also significant factors in toxicity.

In addition, results from experiments with animals may not be readily applicable to humans. Experimental animals such as rats and mice have a much shorter life-span than man and react and respond differently to chemicals because of their own distinctive physiological processes.

Table 151 gives a comprehensive summary of the presence of metals in the environment, their toxicity to humans, and their half-life in the body (the time it takes for one-half the chemical to be excreted). Five heavy metals -- cadmium, lead, mercury nickel as nickel carbonyl, and beryllium -- represent known hazards to human health. Lead, mercury, and cadmium are particularly insidious,because they can be retained in the body for a relatively long time and can accumulate as poisons. Antimony, arsenic, cadmium, lead, and mercuric salts are the most toxic. Discussions of fatal doses and other considerations can be found in each individual metal section.

Lead

Except in cases of prolonged exposure at high concentrations, most ingested lead is absorbed into the blood and is later excreted in the urine. The blood lead does not rise to acute levels; however, a small portion of the daily lead intake gradually accumulates in bones, where it is normally insoluble and harmless. Under certain conditions, such as periods of high calcium metabolism in feverish illness, cortisone therapy, or old age, this accumulated lead can be released suddenly into the blood at toxic levels. A fatal dose of absorbed lead has been estimated to be 0.5 g; ingestion of more than 0.5 mg/ℓ/day may, because of the abovementioned accumulation, cause toxicity and death.

TABLE 151. METALS IN THE ENVIRONMENT AND THEIR TOXICITY[a]

Metal	Food and Water (ppm)	Air (ppm)	Oral dose producing toxicity (mg)	Fatal dose (ingestion)	Total Body content (mg)	Whole body half-life (days)
Antimony	100	1.7	100	100-200 mg	7.9	38
Arsenic	400-900	--	5-50	120 mg	15-20	280
Barium	735	30	200	1 g	22	65
Beryllium	12	0.04	--	--	0.3	180
Cadmium	20 to 100	7.4	3	--	50	25 yrs
Chromium	245	1.1	200	5 g	1.8	616
Copper	1,325	11.4	50-250	10 g[b]	72	80
Iron	15,000	84	--	5-10 g[c]	4,200	800
Lead	300	46	--	0.5 g	120	1,460
Manganese	4,400	28.8	--	--	12	17
Mercury	25	--	--	20 mg - 1 g[d]	--	70[e]
Nickel	600	2.36	--	--	10	667
Selenium	62	--	5	--	14.6	11
Silver	60-80	--	60	2 g[f]	1	5
Tin	7,300	0.6	2,000	--	17	35
Zinc	14,500	16.8	--	10 g[g]	2,300	933

[a] Data primarily from Dreisbach (343) and Oregon State University (1575)

[b] Copper sulfate

[c] A two-yr-old child

[d] Mercuric salts

[e] Methyl mercury

[f] Silver nitrate

[g] Zinc/sulfate

Lead prevents the formation of hemoglobin in the blood by interfering with the synthesis of certain precursors (prorphyrins). This leads to the anemia present in chronic lead poisoning. Lead also inhibits the sulfhydryl enzymes that catalyze many of the metabolic pathways, including the biosynthesis of heme. The effects of lead on the brain and peripheral nervous system are most serious, manifested as gastrointestinal or central nervous system disturbances and anemia. Symptoms of acute poisoning include metallic taste, abdominal pain, vomiting, diarrhea, black stools, oliguria, collapse, and coma. Symptoms of chronic poisoning in early stages include loss of appetite and weight, fatigue, headache, lead line on gums, loss of recently developed skills, and anemia. In advanced stages, there is intermittent vomiting, irritability; nervousness; lack of coordination; vague pains in arms, legs, joints, and abdomen; paralysis; disturbances of menstrual cycle; and abortion. Exposure to tetraethyl lead or tetramethyl lead causes insomina, disturbing dreams, emotional instability, hyperactivity, and even toxic psychosis. Severe symptoms include persistent vomiting, papilledema, encephalopathy (any disease of the brain), elevated blood pressure, cranial nerve paralysis, delirium, convulsions, and coma (343).

There are up to 100 cases of lead poisoning reported annually; an average of 10 are fatal. Most of the fatalities are related to children who ingested lead-based paint from homes built before 1940; however, 7 cases of lead poisoning were reported from drinking well water in Australia in 1973 (104, 270). The well water contained a soluble lead content of about 14 mg/ℓ.

Mercury

Neither methyl nor elemental mercury is normally found in dangerous concentrations in air, water, or most common foodstuffs. There are three principal ways in which man can be poisoned by methyl mercury: (1) when food is consumed that has been contaminated with methyl mercury, e.g., seed containing mercury fungicides ($HgCl_2$); (2) when methyl mercury used or formed in industrial processes is intentionally or unintentionally dumped into natural waters, reaching man directly through the water and indirectly through the food chain; and (3) when nontoxic inorganic or organic phenyl mercury is converted into toxic alkyl mercury compounds by microorganisms in the environment and is passed on to man through the food chain (644).

The acute toxicity of methyl mercury is the result of almost complete (98 percent) absorption of the compound from the gastrointestinal tract. Ingested metallic mercury is not toxic since it is not absorbed. Mercurous chloride and organic mercurials such as acetomeroctol, ammoniated mercury, merbromin, mercocresol, and mercury protoiodide are not likely to cause acute poisoning because they also are poorly absorbed. The single fatal dose of these compounds is 3 to 5 times the fatal dose of soluble mercury salts. The mercurial diuretics (mersalyl, meralluvide, mercurophylline, mercumatilin, mercaptomerin, chlormerodrin, and merethoxylline) are almost as toxic as mercury salts. Volatile diethyl and dimethyl mercury are 10 times as toxic as mercuric chloride (1503). The fatal dose of mercuric salts is 20 mg to 1 g. The biological half-life of methyl mercury compounds is estimated to be about 70 (30 to 100) days (1323, 1586).

Acute poisoning by ingestion of mercuric salts causes metallic taste, thirst, severe abdominal pain, vomiting, and bloody diarrhea. Death is from uremia (an excess of urea and other nitrogenous waste in blood). Ingestion of insoluble or poorly dissociated mercuric salts (including mercurous chloride and organic mercurial compounds) over a prolonged period causes urticaria progressing to weeping dermatitis, stomatitis, salivation, diarrhea, anemia, liver damage, and renal damage progressing to acute renal failure with anuria (total suppression of urine). In children, repeated administration of calomel (Hg_2Cl_2) appears to be the cause of a syndrome known as erythredema polyneuropathy. In fatalities from mercury poisoning, the pathologic findings are acute tubular and glomerular degeneration or hemorrhagic glomerular nephritis. The mucosa of the gastrointestinal tract shows inflammation, congestion, coagulation, and corrosion (343).

Mercurialism is manifested primarily in kidney, liver, or brain damage in animals. Exposure to inorganic mercury compounds usually results in kidney damage, while alkyl mercurialism is characterized by brain damage. However, some degree of both kidney and neurological injury results from exposure to either category of mercurials. Mercury poisoning apparently damages Kreb's cycle enzymes (which catalyze the oxidation of tricarboxylic acids) and protein synthesis, leading to kidney and brain damage.

The tragedy of Minamata Bay, Japan, which occurred from 1953 to 1961, is one of the best documented cases of mercury poisoning. In essence, it was concluded that the disease -- which had felled many inhabitants of the fishing villages on the shores of Minamata Bay -- had resulted from mercury poisoning. A chemical plant in the area used mercury chloride as a catalyst in the production of vinyl chloride. The waste containing the mercury washed off the product was discharged into the bay, and was ingested by the shellfish therein. Some 114 cases of "Minamata disease" were reported, as well as 44 deaths and 22 cases of brain damage in 400 live births (625, 780, 863).

A similar disaster struck at Niigata, Japan, where 120 persons were poisoned (780, 863).

Nickel

The fatal dose of nickel is not known, but its whole body half-life is about 667 days (as shown in Table 157). Inhaled nickel carbonyl decomposes to metallic nickel, which deposits on the epithelium of the lung. This finely divided nickel is rapidly absorbed and damages the lung and brain. The principal manifestation of nickel carbonyl poisoning is dyspnea (difficult respiration). Workers exposed to nickel carbonyl show a high incidence of lung cancer, and some workers develop dermatitis (343). Very little is understood about the adverse health effects of nickel in waste or water supplies.

Cadmium

Cadmium has become the most recent menace among the widely used heavy metals. Unlike lead and mercury, relatively little is known about the fate and distribution of cadmium in the environment, and what is known is often contradictory.

There is very little information available concerning the health hazards of cadmium present in wastewater and water supplies. However, the daily average American citizen's intake of cadmium from foods and water supplies is estimated to be between 0.02 and 0.1 mg/ℓ. The oral dose of cadmium producing toxicity is about 3 mg, but its fatal dose is not known. The whole body half-life of cadmium is 25 yrs (433).

Currently, there is no agreement as to whether cadmium is linked to hypertension in man. Some of the conflicting data are discussed in Friberg's work (433). Circumstantial evidence points to some link between trace metals of cadmium and hypertension: levels of cadmium or ratios of cadmium to zinc were much higher in the kidneys of persons dying from hypertension-related diseases (1323). Inhalation of tobacco smoke is a major source of cadmium accumulation in man. Only about 2 percent of the cadmium ingested through food or drink is absorbed by the body, while 10 percent to perhaps as much as 50 percent of inhaled cadmium is retained (1323).

Cadmium tends to accumulate in liver and kidney tissues because of its very long biological half-life in man (estimates range from 10 to 25 yr, compared with about 70 days for methyl mercury). Excessive levels in the kidney cortex (over 200 ug/g wet weight) result in proteinuria. Therefore, the cadmium concentration in water must be kept low (1323). The Environmental Protection Agency (EPA) in its 1975 Interim Primary Drinking Water Standards set a mandatory limit of 0.010 mg/ℓ for cadmium concentrations in drinking water; the World Health Organization set a limit of 0.05 mg/ℓ. The results of a U.S. geological survey investigation of 720 waterways showed that 4 percent had concentrations above EPA standards (863).

Cadmium has reportedly caused a number of deaths from orgal ingestion of the metal in food or water. For example, Japanese people living along the Jintsu River suffered for years from an unknown malady characterized by kidney malfunction, a drop in the phosphate level of the blood serum, loss of minerals from the bones, and osteomalacia (resulting in pathologic bone fractures causing intense pain). Implicated as one of several sources of the malady was a cadmium, zinc, and lead mine that was discharging wastewater into the river. The disease, known as itai itai, was contracted either by drinking water from the river or by eating rice that had accumulated the metal from irrigation water (863). In another situation, there was an outbreak of acute gastroenteritis in 13 children who drank orange soda contaminated with cadmium at a concentration of 16 mg/ℓ. The contamination was caused by the orange soda coming in contact with the soldered joints in the tank of the soft drink machine (104, 111).

Chromium

Chromium, which exists in various oxidation states (+2+3 and +6), appears to be most toxic to man as the hexavalent chromium ion. The fatal dose of a soluble chromate such as potassium chromate, potassium bichromate, or chromic acid is approximately 5 g. Acute poisoning from ingestion is manifested by dizziness, intense thirst, abdominal pain, vomiting, shock, oliguria (scanty urination), or anuria. Hemorraghic nephritis occurs, and death is from uremia.

Repeated skin contact with chromium leads to incapacitating eczematous dermatitis with edema and slowly healing ulceration. Breathing chromium fumes over long periods of time causes painless ulceration, bleeding, and perforation of the nasal septum accompanied by a foul nasal discharge (343).

Whether chromium is carcinogenic is questionable at this time. However, the incidence of lung cancer in workers exposed to dusty chromite, chromic oxide, and chromium ores is reported to be up to 15 times the expected rate (343).

Experiments on rats showed no toxic response from drinking water containing 0.45 to 25 mg/ℓ in chromate and chromium ion form (831).

Although a potential health hazard exists, evidence of health problems resulting from chromium present in wastewaters is lacking in the literature.

Arsenic

Arsenic is widely distributed in nature. It is present in toxic concentrations in many water supplies: cattle in New Zealand have died from drinking water containing natural arsenic, and there are several areas of the world where there is a high incidence of skin cancer among people drinking well water that contains natural arsenic (863). In 1971 the U.S. Geological Survey found that 2 percent of the samples drawn from 720 waterways were above their standard of 0.05 ppm for arsenic.

Before the advent of modern insecticides, arsenic compounds were widely used to treat food crops. Although arsenic can stimulate plant growth in very low concentrations, in excessive quantities, as little as 1 ppm of arsenic trioxide can be injurious. Organic arsenicals, such as arsphenamine and dimethylarsinic acid, release arsenic slowly and are less likely to cause acute poisoning. But arsenic accumulates in the body, so small doses can be lethal. Repeated or prolonged intake has a cumulative toxic effect, presumably caused by the arsenic combining with sulfhydryl (-SH) enzymes and interfering with cellular metabolism.

Chronic poisoning from ingestion or inhalation of arsenic can cause anemia, weight loss, polyneuritis, optic neuritis, dermatitis, cirrhosis of the liver, abdominal cramps, chronic nephritis, and cardiac failure (343).

Arsenic is suspected to be carcinogenic but not tumorigenic (66, 679, 733). Chronic exposure to arsenic-contaminated water of 0.3 mg/ℓ is also suspected to be related to the increased incidence of hyperkeratosis (hypertrophy of the horny layer of the epidermis) and skin cancer (1569); chronic exposure at levels of 0.8 mg/ℓ may be related to gangrene of the lower limbs (503).

Arsenic is one of the impurities in mineral phosphate deposits, a source of commercial water softeners. Concentrations of 10 to 70 ppm of arsenic have been detected in several common household detergents. Baby rash, hand rash, skin eruptions, and other types of dermatitis allergies are associated with arsenic in detergents. Much of the sewage containing such detergents is dumped into waterways (363). There is a danger that arsenic in laundry water may be absorbed through unbroken skin.

The principal oxidation states of arsenic are +3 and +5. In the +3 state, arsenic forms arsenious oxide; in the +5 state, arsenious acid; and in the -3 state, arsine gas. The fatal dose of arsenic trioxide is about 20 mg. Arsenic poisoning is manifested by gastrointestinal disturbances; death is due to circulatory failure as a result of hemolysis (destruction of the red blood cells).

Copper

The biological properties of copper are such that it is a useful biocide, especially as an algicide. Various salts of copper (one of the most common being copper sulfate - $CuSO_4$) are used as astringents, deodorants, and antiseptics. These salts are all water soluble, and their protein-precipitating characteristics form the basis of their astringent and antiseptic effects. The copper mined in the United States for these effects alone amounts to some 15 million lb/yr and accounts for about 40 percent of all chemical uses of the metal.

Trace amounts of copper are essential for normal metabolism, and relatively large concentrations can be tolerated by most animals including vertebrates. The effect of copper on aquatic organisms varies greatly; microorganisms, including algae, are highly susceptible.

Small quantities of copper are not considered toxic, but higher concentrations are known to cause vomiting and liver damage (1586). Fatalities have been reported following the ingestion of 10 g of zinc or copper sulfate (343). An outbreak of acute copper poisoning occurred in a school in Mesa, Arizona. The outbreak began 10 min after the students drank an orange-flavored drink that had been kept in a brass container for 17 hr (104). An 8-oz glass of the drink contained 8.5 mg of copper.

Copper salts in natural waters occur in small concentrations, from a trace to about 50 ppb. Their presence in injurious amounts is almost always due to pollution.

Selenium

Selenium is an excellent example of the fine balance that can occur in nature between the beneficial and injurious effects of a natural substance. An essential micronutrient for some plants, selenium is one of the most toxic substances to occur naturally in the environment; concentrations only slightly above those needed for growth of plants may be poisonous to animals. A selenium compound that was used to kill insect pests in fruit orchards left a residue in the fruit. Cattle that were feeding on this fruit contracted a chronic form of livestock poisoning called alkali disease. Ingestion of forage containing about 25 ppm can cause this disease when the forage is eaten for several weeks or months (863).

As long ago as 1936, inhabitants of seleniferous areas of South Dakota and Nebraska complained of gastrointestinal symptoms and were found to excrete large quantities of selenium in their urine (863). There is also a report of selenium toxicity from a three-month exposure to well water containing 9 mg/ℓ of selenium.

The American Conference of Governmental Industrial Hygienists recommends a selenium limit of 0.2 mg/cu m in air and 0.01 ppm in water.

Beryllium

Although the fatal dosage of beryllium is not known, in 1971 the EPA placed beryllium on the list of hazardous pollutants. From 1941 to 1967, 760 cases of berylliosis were recorded (343).

Soluble beryllium salts are irritating to the skin and mucous membranes, and induce acute pneumonitis with pulmonary edema. At least part of the changes present in acute pneumonitis and berylliosis (chronic pulmonary granulomatosis) develop from hypersensitivity to beryllium in the tissues. Weight loss and marked dyspenea, which are symptoms of berylliosis, begin 3 months to 11 years after the first exposure (inhalation). Eczematous dermatitis with a maculopopular, erythematous visicular rash appears in a large percentage of workers exposed to beryllium dusts.

Barium

Absorbable salts of barium, such as the carbonate, hydroxide, or chloride, are used in pesticides; the sulfide is sometimes used in depilatories for external application. A soluble barium salt such as the carbonate or hydroxide may be present as a contaminant in the insoluble barium sulfate used as a radiopaque contrast medium.

The fatal dose of absorbed barium is approximately 1 g. The principal manifestations of barium poisoning are tremors and convulsions. Barium presumably induces a change in permeability or polarization of the cellular membrane that results in stimulation of all cells indiscriminantly (343).

Other Elements

Antimony will be considered a potential hazard if levels of the element increase. It is present chiefly in industrial wastes, typesetting metal, pewter, and enamel ware. Antimony is suspected to cause a shortened life-span and heart disease in rats (343).

Metal alloys and smoke suppresant in power plants are the major sources of manganese as a pollutant. Although manganese in trace amounts is an essential element to man, increased levels may imperil health. The findings in one death - suspected to be caused by the ingestion of manganese-contaminated drinking water - were atrophy (a wasting of the tissues) and disappearances of cells of the globus pallidus (in the brain). Experimental animals show inflammatory changes in both gray and white matter (343).

BIOCIDAL CONTAMINANTS

There is significant information in the literature concerning biocidal contaminants, their chemical and physical characteristics, toxicology, analytical chemistry, and impacts on health and the environment. In both chemical and medical literature, hundreds of cases of acute poisoning resulting directly or indirectly from such pesticides have been reported (see Table 152).

Chlorinated Hydrocarbons

Because of their persistence and nondegradational characteristics, many of the relatively less toxic chlorinated hydrocarbons, such as DDT, aldrin, and dieldrin, have been banned and are being replaced by highly toxic but less persistent organophosphorus pesticides. The U.S. EPA is presently conducting a program to identify new, less harmful

TABLE 152. BIOCIDES IN THE ENVIRONMENT AND THEIR TOXICITY*

Biocide	LD_{50} (oral) mg/kg	Fatal Doses (g/kg)	Chronic Poisoning	Acute Poisoning
DDT	Rat 285 Rabbit 325	0.4	Not substantiated. Having 648 ppm in their body remained well.	Severe vomiting within 30 min to 1 hr of 5 g. Weakness and numbness of the extremities. Apprehension and excitement are marked.
Dieldrin	Rat 60 Dog 68 Rabbit 45	0.07	Not been established in man. Impair liver function in animals, occasional epileptiform convulsions.	Hyperexcitability, tremors, ataxia, convulsions.
Lindane (Benzene hexachloride)	Rat 135 Dog 120 Rabbit 130	0.6	In animals, liver necrosis.	Vomiting and diarrhea, convulsions, circulatory failure.
Malathion	Rat 2500	0.86	In animals, colinesterase levels of red blood cells and plasma are reduced markedly.	Headache, tremors, nausea, abdominal cramps, diarrhea, coma, heat block.
Parthione	Rat 4	0.0014	Not established in man.	Similar to those of malathione, but more severe and fatal.

TABLE 152. (continued)

Biocide	LD_{50} (oral) mg/kg	Fatal Doses (g/kg)	Chronic Poisoning	Acute Poisoning
2, 4-D	Mouse 375	0.7	Weakness, fall of blood pressure, muscle damage.	Burning pain, painful and tender muscle, fever, paralysis, irreversible fall of blood pressure.
2, 4, 5-T	Rat 300 Dog 100	0.6	Similar to 2, 4-D	Similar to 2, 4-D.

* Data primarily from Sunshine (1583), Dreisbach (343), and McKee and Wolfe (1579).

pesticides that can act as substitutes for those causing the problems.

This decision to remove aldrin and dieldrin from the market has been highly criticized. The idea that these compounds pose "an unreasonable risk of cancer" is based on the presence of tumors in the livers of mice that were given food containing aldrin and dieldrin; some of these tumors metastasized to the lungs. However, it has been noted that similar tumors can be produced by other compounds, such as DDT and phenobarbitone, and can occur in mice on normal diets. There is, then, some question as to whether the production of tumors in the livers of mice given aldrin or dieldrin is a reliable indicator of a hazard to man. The general dietary intake of all major pesticides except aldrin and dieldrin is well below the allowable standards that were established in 1972 by the World Health Organization Council on Environmental Quality. Much of the DDT uptake (about 85 to 90 percent) comes from food; the remainder comes from air, water, aerosols, cosmetics, and clothing.

Because of DDT's extremely low solubility in water (about 2 ppb), the body, which is essentially a water system, cannot handle the liquid soluble substance and deposits it in fat. Quantities of DDT and other related pesticides do not appear to build up continuously; instead, the pesticides reach a plateau or steady state (storage equilibrium) at which they are being excreted and degraded at levels equal to their intake. Children may take 5 to 10 yr to reach storage equilibrium. Little is really known about these levels, which undoubtedly vary with different pesticides, exposure, intake conditions, and individuals.

Of real concern are the possible effects of long-term exposure to low levels of DDT and other pesticides. Some sublethal effects have been observed in animals (cellular changes in liver tissue and other physiological and histological effects); however, these effects cannot be extrapolated to man. Of course, there is still the suspicion that DDT might eventually cause damage to human physiology. It has been suggested, for instance, that long-term exposure to low levels of pesticides may cause cancer. Research neither supports nor contradicts this possibility, because it is difficult to control or document experiments on humans over a longer period of time.

Based on animal experiments, the average lethal amount of DDT is a single dose of about 8,000 to 14,000 mg/150-lb person, although quantities as high as 109 ppm have been found in sampling a general population. A study was conducted of persons accidentally or violently killed in Dade County, Florida, from 1965 to 1967. The average concentration of

DDT in the fat of the bodies ranged from 5 to 22 ppm; there was more in adults than in children, more in nonwhites than in whites.

Because the use of chlorinated hydrocarbon pesticides has been sharply curtailed, it is therefore expected that smaller quantities of DDT are being ingested and that residues in human fat will decline. However, the average value of DDT, DDE, and isomers in human fat samples in India was 21.8 ± 2.9 mg/kg in 1973 and 24.3 mg/kg in 1965. This would seem to indicate that the DDT storage status had not undergone any significant change in those years.

A comparative study was performed of DDT and its derivatives in human blood samples. The study was conducted in two areas of Ontario where DDT has been used in large amounts. The mean values of total DDT in the adipose tissue and blood samples were found to be 5.83 and 0.032 ppm, respectively. There was a significant correlation between total DDT in the fat and blood, but there was no evidence of any adverse effects of DDT on either of the populations (150).

In another study, tap water samples collected in the Washington, D.C., area before treatment showed the presence of DDT (0.17 mg/ℓ), TDE (0.27 mg/ℓ), and DDT-derived compounds (.15 mg/ℓ); no evidence of these compounds was found after treatment.

The EPA's ban on DDT has been modified since the institution of the ban in 1972. In 1974, DDT was used against the tussock moth in northwestern forests and against the pea leaf weevil in Washington and Idaho. The EPA, however, states that DDT will be used only against pests that cannot be controlled by other means and that will do economic damage or threaten human health or safety. Also, despite the suspension of aldrin and dieldrin, the EPA will permit their use against termites and clothes moths, under certain circumstances.

The toxic effects of carbaryl, which is used against DDT-resistant lice, were found to be minimal at the levels of 75 mg/ℓ (256).

The residues of mirex - a chemical which is used to control the imported fire ant - were discovered in the fatty tissue of six persons from 1971 to 1972. It was concluded that the residues came from routes other than direct pesticide usage (994).

The maximum allowable concentration of most of the pesticides (as determined by biological tests) lies below 1 mg/ℓ, but is as low as about 0.01 mg/ℓ for such preparations as atrazine, malathion, and thiometon (1096).

Workers occupationally exposed - through manufacturing processes - to aldrin, dieldrin, endrin, and telodrin (isobenzan) for up to 15 yr were studied by Versteeg and Jugar (1431). No persistent adverse effects on health were observed in 52 workers who had left the company and were traced. The average pesticide exposure in this group of workers was 6.6 yr, and an average of 7.4 yr had elapsed since exposure. No hepatic disease or convulsions occurred, and no new cases of malignant disease developed. Most of the workers experienced no unusual illnesses.

Exogenous leukemogenic agents such as pesticides (DDT, lindane, organophosphates) are being considered as potentially contributory etiological factors that may activate a latent leukemia virus in certain cells (1329).

Organophosphorus Pesticides

The organophosphates attack the neural transmission system of mammals and arthropods and thus interfere with the function of the target organs. Especially active are those organophosphates that function by inhibiting carboxylic ester hydroxylases (including acetylcholinesterase, which is present in human erythrocytes, nerves, synapses, and skeletal muscle; and cholinesterase, which is present in human plasma and in the liver). Miosis (contraction of the pupil), is found in about 90 percent of the patients with moderately severe or severe cases of organophosphate poisoning (1521).

Tamura et al. (1358) conducted a statistical and epidemiologic investigation of the relationship between the changes in the use of organophosphorus insecticides and the recent increase in cases of myopia in 40,000 Japanese children from 1957 to 1973. The study showed that in the year following any period when large amounts of organophosphorus insecticides were used, the morbidity from myopia in school children increased rapidly; conversely, decreased use of the insecticides resulted in decreased morbidity. Thus it appears that the recent increase in myopia in school children was due, at least in part, to chronic intoxication or organophosphorus insecticides (1358).

Pediatric hazards associated with organophosphates are often reported (1578). Signs of toxicity are overaction of the parasympathetic nervous system, nausea, vomiting, diarrhea, sweating, and abdominal cramps. Large doses may lead to muscular paralysis and death from respiratory failure (1578).

Because organophosphorus pesticides are readily absorbed through the skin, as well as by ingestion and inhalation, these pesticides present a particular hazard to agricultural workers who engage in mixing, loading, and applying the concentrated materials (1309).

The incidence of organophosphorus insecticide poisoning varies considerably throughout the world. During 17 yr in Japan there were 19,000 cases of organophosphorus poisoning reported, resulting in more than 9,000 deaths. By contrast, during 17 yr in Great Britain there were only five deaths reported from this poisoning (1415). Increased chromosome aberrations were observed in patients suffering acute organic phosphate insecticide intoxication. The frequency of stable chromosome aberrations showed a significant increase with malathion, trichlorfon, mevinphos, and methylparathion; malathion induced an outstandingly high number of structural chromosome aberrations. Patients less seriously intoxicated suffered milder chromosome alterations (1585). Even in the absence of clinical signs of organophosphate poisoning by dichloros at low levels, the cholinesterase values were much lower than those of a nonworker control population. The decrease in enzyme levels was significantly correlated with subjective health complaints: sore throat and loss of memory (1567).

Herbicides

Today over 40 weed killers are available, but the most widely used are 2, 4-D (2, 4 - dichlorophenoxyacetic acid) and 2, 4, 5-T (2, 4, 5 - trichlorophenoxyacetic acid). In general, these chemicals are rarely lethal to humans or animals, do not persist for long periods of time in the environment, and do not build up in the food chain.

During the Vietnam war, South Vietnam suffered massive military herbicide spraying. It has been estimated that at least one-third of the timber forests was destroyed, as well as 10 percent of all cultivated land and at least 25 percent of the coastal Mangrove forests, which are the breeding or nursery grounds for most offshore fish and crustaceans (1485). However, there was no evidence in Vietnamese hospital records that were examined by Thimann (1372) that birth defects could be attributed to the herbicide spraying.

Humans are exposed minimally to the phenoxy herbicides through food; air and water are the primary sources of exposure to these herbicides. On the basis of air samples collected from wheat-growing areas in the state of Washington, it was estimated that an average person would be exposed to 1.8 mg of phenoxy herbicide/day. Rain and wind carry the herbicides into water (215). Workmen who were engaged in the manufacture of herbicides were examined; the clinical results were compared to those of a control population that was not exposed to the 2, 4-D or 2, 4, 5-T. No meaningful differences were obtained; moreover, there were no chromosomal effects (215).

In the fall of 1969, the National Institute of Cancer reported that 2, 4, 5-T was teratogenic and was responsible for fetal toxicity at levels of 27 ± 8 ppm. The manufacturer claimed that this herbicide was not teratogenic; that the fetal toxicity was caused by 2, 3, 7, 8-tetrachlorodibenzo-p-dioxin, known as dioxin (653).

At the dose level of 100 mg/kg/day of silvex, there were no effects on either the dams or fetuses of rats. Similar observations were also made with picloram. Picloram, however, is much more persistent in soil than the other compounds. Unlike DDT, it forms soluble salts and does not concentrate in fat; in addition, the toxicity of picloram is very low when compared to that of most other herbicides (653).

The widespread use of phenoxy herbicides has produced no demonstrable evidence of potential harm to man. The herbicides used most widely (2, 4-D and 2, 4, 5-T) are degraded to nontoxic components and do not bioconcentrate (653).

An epidemiological investigation was made of tumor incidence and mortality in Swedish railway workers exposed to various herbicides. Among workers exposed to amitrole (3-amino-1, 2, 4-triazole), tumor incidence and mortality were significantly increased and were slightly dose related. Nearly normal conditions were found in those exposed to phenoxy acids. Animal experiments suggest that amitrole may produce malignant tumors in several different organs, but tumors of the thyroid and liver have received the most attention (48).

To receive a letal dose of 2, 4, 5-T, a 125-lb woman would have to eat the equivalent of her body weight of material containing 380 ppm of the herbicide (based on the acute oral toxicity expressed in mg/kg). Both 2, 4, 5-T and DDT in technical form are relatively nontoxic on skin contact (3,800 mg/kg for 2, 4, 5-T and 2,500 mg/kg for DDT). There have been no human deaths and remarkably few human illnesses from the agricultural or public health uses of either of these chemicals. A no-effect level of 50 mg/kg/day of 2, 4, 5-T containing less than 1 ppm dioxins has been proposed as providing ample protection for human embryos (215).

The first recognized incident in which significant poisoning resulted from the improper disposal of waste residues containing TCDD (tetrachlorodibenzodioxin) was in eastern Missouri. In this incident, a salvage oil company sprayed waste-oil sludge on an arena at a horse-breeding farm, to control dust. Birds, cats, dogs, and rodents were killed; 62 out of 85 horses became ill, and 48 died. The horses had been exposed to the arena in the summer of 1971, and they continued to die as late as January 1974. The horses showed chronic weight loss, loss of hair, skin lesions, dependent edema, intestinal colic, dark urine, gross hematuria (the passage of blood in the urine), conjunctivitis, joint stiffness, and laminitis (inflammation of one side of the neural arch of a vertebra). In addition, the horses' feet were inflamed. Human illness, although less severe, included one case of hemorrahagic cystitis in a 6-yr-old girl who played in the arena. The soil of the arena contained 3.18 to 3.3 percent TCDD by weight and 2, 4, 5-tri chlorophenol (TCP) and related chemicals (1431).

Paraquat, a widely used bipyridyl herbicide, produces a low-dose, chronic illness in rats, primarily manifested by pulmonary fibrosis (414). Paraquat is also known to cause severe lung damage in rats, leading to death (599). Although it causes proliferation of fibroblasts, no carcinogenic action has been demonstrated. Taken orally, paraquat causes ulceration in the digestive tract, diarrhea and vomiting, renal damage, and jaundice.

Pesticidal Viruses

Tinsley and Melnick (1387) found some antibodies that reacted with insect viruses in domestic and wild animals and in several laboratory workers handling insect viruses. There is always a possibility that changes in the pathogenicity and specificity of pesticidal insect viruses could occur, causing a wider spectrum of host involvement. No collaborative research programs on the in vitro specificity of insect viruses were recommended (1387).

Fungicides

In Iraq during a two-month period, 6,530 poisoning victims were hospitalized, and 459 hospital deaths occurred. The source of the poisoning was found to be homemade bread prepared from seed wheat treated with methyl mercurial fungicides (1358). In another case, diphenyl (or biphenyl) poisoning from fungicides in a Finnish paper mill was reported by Seppäläinen and Häkkinen (1235). Workers were employed in areas of the mill where the average concentration of diphenyl measured in the air varied from 0.6 to 123.0 mg/m^3. The workers developed EEG abnormalities that were compatible with generalized cerebral disturbance (1235).

Polychlorinated Biphenyls (PCB's)

Relatively high concentrations of a group of widely used industrial chemicals known as PCB's have been found in fish, birds, and man. The widespread presence of these PCB's has tagged them, like DDT, as truly global pollutants. In fact, it has been speculated that sunlight might convert DDT to PCB's. PCB's are insoluble in water, soluble in fats and oils, and very resistant to chemical and biological degradation. Due to their solubility, PCB's accumulate in the environment, especially in aquatic organisms and birds (in which cumulation factors up to one billion may be reached). PCB's, like DDT, can inhibit photosynthesis of marine phytoplankton and can kill shrimp, trout, minks, and birds. PCB's may be twice as effective as DDT in causing thinning of bird eggshells.

In 1968, over 1,000 people in Japan suffered from a skin disease and from liver damage caused by rice oils. The oil was heavily contaminated with PCB's; however, these effects were not due to the PCB's but to a highly poisonous contaminant - chlorinated dibenzofurans. This contaminant is found in some PCB's manufactured in other countries.

To date limited study in this area indicates that uncontaminated PCB's have a very low toxicity to man. According to the Food and Drug Administration, the average PCB concentration in a normal American diet is only about 10 percent of the strict safety levels set in 1971 for food, food packaging materials, and animal feeds.

SYNTHETIC/ORGANIC CONTAMINANTS

Recently diverse compounds identified in water supplies drawn from the Mississippi River have been discovered in the blood serum of local residents using the water supply. This has created great concern over chemicals found in drinking water. The presence of small amounts of synthetic/organic chemicals in treated reclaimed water has been recognized as a potential health hazard.

The list of compounds identified in drinking waters is rapidly growing larger. This is due primarily to the continual introduction of new chemicals but also to the development of sensitive analytical techniques that measure trace quantities of the chemicals. There is very little evidence available concerning the relation between the presence of these compounds in water and human disease. Information on classical acute health effects of relatively toxic chemicals can be obtained from physicians' manuals. However, knowledge of the chronic health effects associated with long-term exposure to low-level concentrations of chemical substances is not well documented. The possibility that cancer may result from long-term exposure to low concentrations of carcinogens is of utmost concern.

Carbon-Chloroform Extractables (CCE) and Carbon Alcohol Extractables (CAE)

The Committee on Water Quality Criteria (1586) suggested that absorbable organic carbon in public water supply sources should not exceed the carbon chloroform extractables (CCE) level of 0.7 mg/ℓ. (No level has been established for carbon alcohol extractables - CAE.) The establishment of this level was based upon the adverse physiological effects of CCE as well as aesthetic considerations.

To date, laboratory testing of the pathological effects of trace organics has been restricted to mice and fish. Hueper and Payne (609) conducted a study of mice that were exposed subcutaneously, cutaneously, and orally to extracts of CCE and CAE obtained from both raw and finished water supplies. Results of this study indicated that these extracts had a potential for carcinogenicity (see Table 153). The cutaneous dose used in the experiment was one drop of extract every 2 weeks for 56 weeks with 72 mice; the subcutaneous dose was 2 to 4 mg every 2 weeks for 56 weeks with 72 mice; and the oral dose (only one instance) constituted 2 percent of the regular food and was available to 40 mice. The control group consisted of 40 unexposed mice. No tumors were observed among the 40 mice in the control group or among those exposed orally. Carcinogenic effects in subcutaneous and cutaneous groups included one papilloma (a circumscribed overgrowth of the papilla) of the bladder; four spindle-cell sarcoma (fusiform cell tumor) at the site of subcutaneous injection; and leukemia, lymphoma, or reticulum-cell sarcoma of the liver in all other instances. However, it is very difficult to extrapolate the experimental data of animals to humans. Thus, the question still remains as to whether such extracts would cause similar abnormalities in humans.

Another experiment was conducted to investigate the toxicity of CCE and CAE from processed water supplies (355). A total of 5 mg of either CCE or CAE was introduced subcutaneously during the first 20 days after birth of the test mice. No tumors were found to be induced during the

TABLE 153. SUMMARY OF RESULTS OF INTRODUCING CCE AND CAE
FROM RAW AND FINISHED WATER INTO MICE (609)

Route of Exposure	Number of Tumors Produced/Animal Exposed			
	Water Source			
	Raw		Finished	
	CCE	CAE	CCE	CAE
Subcutaneous	5/72	1/72	2/72	1/72
Cutaneous	2/72	1/72	0/72	0/72
Oral	--*	--	0/40	--

* No test.

TABLE 154. LONG-TERM TOXICITY OF TRACE ORGANICS (1570)

Test Fish		Source of Water				Total Accum. Test Time[a] for 50% Kill,d. Test Material Concen., (mg/ℓ)								
Type	Length,(cm) Weight, (g)	Location	Run	Unit	Organic Extract	10		7.5		4.2		1.0		Control
						0[c]	1.0[b]	0	0.75	0	0.42	0	0.10	
Trout	8.2/11.0	Meramec Spring	2	2	CCE & CAE[c]	30	20	37	24	48	29	>54	>54	>54[d]
	5.3/2.2	Missouri River	-	-	CCE	10	5	13	11	19	13	>19	>19	28[e]
						24		13.5		5.6				
						0	2.4	0	1.35	0	0.56			Control
Red Shiners	4.7/1.5	Meramec Spring	2	2	CCE & CAE[c]	30	20	39	30	>65	45			>65[e]

[a] Five d exposure and recovery periods were used.

[b] Test material concentration in recover water, mg/ℓ.

[c] Combined at naturally occurring ratio.

[d] Fish died because of loss of temperature control.

[e] Fish showed signs of disease; test discontinued.

TABLE 155. ACUTE AND PREDICTED LONG-TERM TOXICITY OF TRACE ORGANICS (1570)

Test Fish		Source of Water				Acute TLm[a]; (mg/ℓ)				Long-Term TLm	
Type	Length, (cm) Weight, (g)	Location	Run	Unit	Organic Extract	24 hr	48 hr	96 hr	120 hr	mg/ℓ	at days
Trout	10.3/16.1	Meramec Spring	2	1	CCE & CAE[b]	130	125	95	82	33.1	40
				2		88	75	61	56	3.9	70
				3			no effect up to 180				
	5.3/2.8	Missouri River	-	-	CCE	36	32	28	24	0.44	80
Sunfish	7.2/8.9	Meramec Spring	2	1	CCE & CAE	166	141	115	103	25.4	50
				2		137	121	114	100	47.0	50
	7.7/11.7	Missouri River	-	-	CCE	56	49	45	39	30.1	30
Golden Shiners	6.5/4.1	Meramec Spring	2	2	CCE & CAE[b]	180	171	160	152	141	20
		Missouri River	-	-	CCE	59	52	39	33	7.5	40
Red Shiners	5.0/2.0	Meramec River	1	1	CCE & CAE[b]		no effect up to 240				
	2.8/1.2			2			no effect up to 200				
	5.7/1.3		2	1			no effect up to 305				
				2		195	170	148	120	13.3	50

[a] TLm: Median tolerance limit, or the concentration at which 50 percent of the test animals are able to survive for the specified period of exposure.

[b] Combined at naturally occurring ratio.

period, but varied death rates of the test animals were observed. The studies were conducted over a 1-yr period using New Orleans drinking water supplies (355).

Organohalides

Occurrence and formation of organohalides such as $CHCl_3$, $CHCl_2Br$, $CHClBr_2$, and $CHBr_3$ were reported when water containing organic substances was chlorinated (83, 1176). Of the haloforms, chloroform ($CHCl_3$) was reported as the predominant organohalide, with concentrations ranging from 54 $\mu g/\ell$ to 150 $\mu g/\ell$. The level of risk for chloroform - estimated from consideration of the worst case and for the expected cancer site, such as the liver - might be extrapolated to account for up to 40 percent of the observed liver cancer. The toxicity of chloroform has been well demonstrated in lethal-dose studies.

An LD_{50} value ranging from 89 to 35 mg/kg was observed by Tardiff and Deinzer (1362), when CCE obtained from the Kanawha River in West Virginia was introduced into mice via an interperitoneal route. The difference among these LD_{50} values was shown to be due to the amount of chloroform present in the extract, indicating the toxicity of chloroform. The CAE obtained from the same river showed an LD_{50} of 84 mg/kg. The LD_{50} for the concentrated organics from Cincinnati tap water was shown to be 65 to 290 mg/kg.

The same authors also reported the identification of 60 compounds from drinking water. Of the compounds, 1 was classified as nontoxic; 14, moderately toxic; 16, very toxic; 2, extremely toxic; and 27, unknown (135). However, it is difficult at this time to determine the relationships of the toxicities of these compounds in humans to the level of the compounds present in water and wastewater.

Careful interpretation of the toxicity data, such as the LD_{50} value obtained from concentrated extracts, is necessary when these values are to be used to set toxicity levels in drinking water.

Polynuclear Aromatic Hydrocarbons (PAH)

Occurrence, formation, concentration, activity, carcinogenicity, and degradation of polynuclear aromatic hydrocarbons (PAH) in water are well documented (29). Minimal carcinogenic doses of three of the most potent hydrocarbons in susceptible experimental animals are shown in Table 156.

TABLE 156. THE MINIMAL CARCINOGENIC DOSE FOR THREE OF THE MOST POTENT CARCINOGENIC HYDROCARBONS IN SUSCEPTIBLE EXPERIMENTAL ANIMALS (1564)

Carcinogen	Animal	Least Amount which Caused Cancer*
3,4 - Benzpyrene	Mouse	4.0 μg
3,4 - Benzpyrene	Rat	50.0 μg
1,2,5,6 - Dibenzanthracene	Mouse	2.5 μg
20 - Methylcholanthrene	Mouse	4.5 μg
	Rat	20.0 μg

* Only one dose was administered subcutaneously.

Miscellaneous Organic Compounds

A preliminary experiment was designed to study the toxicity of organic compounds present in a secondary treatment plant effluent. Rats were supplied with filter-sterilized effluent from an activated sludge plant as the sole source of drinking water. Two of the 10 rats developed massive tumors. Also, the exposed female rats developed significantly smaller adrenal glands than the control rats that were provided with the local water supply (1037).

An epidemiologic study on the toxicity of compounds present in drinking water was cited by Andelman and Suess (29). This study indicated fewer cancer mortalities in a London borough that was supplied with well water than in boroughs supplied with river water. This could mean that the river water receives more carcinogenic waste material.

Similar findings reported by Tromp (1584) showed that areas using municipal water systems had lower cancer death rates than those using other systems. However, the cancer death rate was higher among areas that received municipal water from a river than among those that received municipal water from wells.

When other factors such as food, air quality, and individual habits (i.e., cigarette smoking) are considered, the importance of trace carcinogens in water supplies may not be significant. However, several observations have been made correlating water supply quality and cancer incidence. For example, Talbot and Harris (1357) established a correlation between cancer mortality in white males and water supply source, between mortality and urbanization, and between mortality and income. When occupational variables are not considered, lung cancer mortality rates were found to be correlated with surface water sources, but there were no correlations found in other cancers.

Halogenated Hydrocarbons

In a study that was designed to investigate the correlation between levels of halogenated hydrocarbons in New Orleans drinking water and levels of halogenated hydrocarbons in blood plasma of individuals drinking the water, 13 halogenated hydrocarbons were isolated. Researchers also detected the presence of carbon tetrachloride and tetrachloroethylene (2 of the 13 compounds) in the pooled sera of eight people. It is probable that biomagnification was involved if the chemicals in the plasma originated from the drinking water (1573).

Low Molecular Sulfurated Hydrocarbons

A case history study of the waterborne goitrogens and their role in the etiology of endemic goiter was recently reported from Colombia, South America (442). The potential presence of low molecular weight compounds (less than 220), such as sulfurated hydrocarbons in water and wastewater, received careful evaluation. The compounds (sulfurated hydrocarbons) were known to be related to the high incidence of goiter among children and were regarded as waterborne goitrogens. The study also reported a 10-fold increase in cancer of the thyroid where endemic goiter was observed (442).

BIOLOGICAL CONTAMINANTS

Epidemics of waterborne diseases have largely been eliminated, due mainly to the advancement of sanitary engineering, enforcement of public health regulations, and preventive medical practices; however, waterborne disease data from the last three decades indicate that outbreaks are no longer on the decline in the United States (271, 425). As shown in Table 157, during the 25-yr period from 1946 to 1970, there were 358 recognized outbreaks (72,358 individuals involved) of disease or chemical poisoning attributed to contaminated drinking water.

TABLE 157. WATERBORNE-DISEASE OUTBREAKS: 1946 TO 1970, BY TYPE OF ILLNESS AND SYSTEM* (308)

Illness	Private Systems		Public Systems		Total	
	Outbreaks	Cases	Outbreaks	Cases	Outbreaks	Cases
Gastroenteritis	121	8,970	57	36,285	178	45,255
Typhoid	47	507	6	103	53	610
Infectious hepatitis	36	1,094	17+	739	53	1,833
Shigellosis	23	1,616	10	5,784	33	7,400
Salmonellosis	7±	118	6	16,612	13	16,730
Amebiasis	4	50	1	25	5	75
Enteropathogenic E. coli	4	188			4	188
Ciardiasis	1	19	2	157	3	176
Leptospirosis			1	9	1	9
Tularemia	2	6			2	6
Poliomyelitis			1	16	1	16
Total	253	12,620	105	59,738	358	72,358

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*The terms public and private as used in this table refer not to the ownership of the utility but to the type of service rendered. Generally, public supplies are municipal-type supplies available to all within a certain area. Private supplies include individual and semipublic sources, such as provided by camps, resorts, hotels, industries, and institutions.

+One gastroenteritis outbreak also included seven cases of infectious hepatitis.

±One gastroenteritis outbreak was preceded by outbreak of 38 cases of salmonellosis.

According to the reports of the Center for Disease Control (426), during the past four years (from 1972 to 1975), 105 waterborne disease outbreaks were reported, involving 22,650 cases. As shown in Table 158, in 1975, 24 waterborne disease outbreaks were reported, involving 10,879 cases.

TABLE 158. WATERBORNE DISEASE OUTBREAKS 1972-1975 F AND W (426)

	1972	1973	1974	1975	Total
Outbreaks	29	24	28	24	105
Cases	1,638	1,720	8,413	10,879	22,650

Table 159 shows the number of outbreaks and cases by etiology and type of water system. The category with the most outbreaks is acute gastrointestinal illness. This category includes outbreaks characterized by upper and/or lower gastrointestinal symptomatology for which no specific etiologic agent was identified. In previous years, these outbreaks were considered sewage poisoning. One outbreak each was caused by giardiasis, shigellosis, enterotoxigenic E. coli, and hepatitis A. There were no reported deaths associated with waterborne disease outbreaks in 1975 (426).

Most outbreaks involved semipublic (67 percent) and municipal (25 percent) water systems; some involved individual (8 percent) systems. Outbreaks attributed to water from municipal systems affected an average of 1,218 persons; those attributed to semipublic systems involved 221 persons; and those associated with individual water systems affected 13 persons. Of those 16 outbreaks associated with semipublic water supplies, 11 (69 percent) involved visitors to areas used mostly for recreational purposes.

TABLE 159. WATERBORNE DISEASE OUTBREAKS, BY ETIOLOGY AND TYPE OF WATER SYSTEM, 1975 (426)

Disease	Municipal		Semipublic		Individual		Total	
	Outbreaks	Cases	Outbreaks	Cases	Outbreaks	Cases	Outbreaks	Cases
Acute gastrointestinal illness	4	7,300	13	2,460	--	--	17	9,760
Chemical poisoning	2	11	1	26	--	--	3	37
Giardiasis	--	--	--	--	1	9	1	9
Shigellosis	--	--	1	56	--	--	1	56
Enterotoxigenic E. coli	--	--	1	1,000	--	--	1	1,000
Hepatitis	--	--	--	--	1	17	1	17
Total	6	7,311	16	3,542	2	26	24	10,879

The object of this section is to compile a comprehensive summary of research into the health effects of biological contaminants in wastewater treatment. Diseases transmitted by water and wastewater largely originate in the intestinal discharge of man and/or animals. These diseases are caused by bacteria, viruses, fungi, and protozoan and other parasites.

Protozoan and Other Parasites

A number of intestinal parasite infections can be introduced into man directly from water supplies and indirectly through wastewater discharges. Under normal conditions, the potable-water route of infection is quite unimportant. However, the reuse of treated waste effluents for irrigation and groundwater recharge requires that this problem be reexamined.

Ascariasis (a disease caused by infection with ascaris), trichuriasis (a disease caused by infection with trichuris), and hookworm diseases are some of the infections that originate from direct soil pollution by feces. This pathway has been virtually eliminated in the United States, due to the introduction of modern sewage disposal and water supply systems. When there is a breakdown in sanitation, these diseases may reappear (1509).

Studies in foreign areas affirm the relative unimportance of public water supplies as a route of infection for intestinal parasites. A study by the World Health Organization (WHO) in

Sudan, in which a modern water supply was provided to one test city, showed that parasitic infections were not decreased. The study indicated, instead, the need for sanitary waste disposal facilities. This finding was substantiated by the substitution of a modern water supply system for an older system in Western Transvaal. The substitution had no effect on the prevalence of helminths among the Bantu population studied - children 7 to 16 years (1509).

Amoebic dysentery (amoebiasis) appears to be the most important parasitical disease associated with wastewater in the United States. It is caused by Entamoeba histolytica, a protozoan. Today, the prevalence rate of E. histolytica in the general population of the United States is considered to be around 3 to 5 percent (1577). The prevalence of the intestinal protozoa varies considerably in different population groups and is generally correlated with socioeconomic conditions. Higher rates are found in areas of poor sanitation and in regions without sewage systems and potable water. Higher rates are also noted in groups of people with poor personal hygiene (e.g., patients in institutions for the mentally retarded).

The amoeba can form small cysts (5 to 20 μm) with a specific gravity of about 1.06. Each mature cyst is capable of producing four motile amoebae. The cysts are resistant to adverse environmental conditions and are excreted with feces into water and/or remain in the human digestive tract to become vegetative amoebae. These amoebae multiply and may become invasive, causing erosion of the superficial mucous membranes. They may eventually invade the tissue with consequent ulceration.

The vegetative forms do not survive outside the digestive tract. As with most parasitic diseases, the symptomatology produced by pathogenic intestinal protozoa is too nonspecific to enable the physician to make an accurate clinical diagnosis. In 1974, there were 2,743 reported cases of amoebiasis in the United States (1580); because of non-clinical manifestations, the actual figure is undoubtedly considerably higher.

In an experiment with volunteers, it has been demonstrated that up to 25 percent could be infected by a dose containing less than 10 organisms of endamoeba; the remainder required a minimal dose of 10,000 organisms to become infected. However, as shown in Table 160, the infected volunteers did not manifest any signs of illness.

Giardia lamblia, a flagellated protozoan of the small intestine, often implicated epidemiologically with drinking water is the etiological agent for giardiasis. An outbreak has recently been reported (1581) in Rome, New York, where the water supply could have been contaminated by untreated human waste. Another

TABLE 160. CLINICAL RESPONSE OF ADULT HUMANS TO VARYING CHALLENGE DOSES OF ENTERIC PATHOGENS (161)

ORGANISM	Challenge doses											
(strain)	10^0	10^1	10^2	10^3	10^4	10^5	10^6	10^7	10^8	10^9	10^{10}	10^{11}
Shigella dysenteriae												
(1) (A-1)			+		++							
(1) (M 131)		+	++	+++	++++							
Shigella flexneri (w)										+++		
(2a)					+	+++	++++	+++	++++			
(2a)			+	+++	+++	+++						
Vibrio cholerae												
inaba 569B (unbuffered)					-		-	-	++	++	-	++1
inaba 569B (+$NaHCO_3$)		-		(+)	+++	+++1	+++1		++1			
ogawa (+$NaHCO_3$)				++			++1					
Salmonella typhi												
(Quailes)vi				-		++		++	++++	++++		
(Zermat)vi					+++							
(Ty2V)vi							++					
(O-901)									++			
(Ty2W)									+			
(Quailes)						+		++		++++		
(Quailes)						++						

TABLE 160. (continued)

ORGANISM (strain)	Challenge doses 10^0	10^1	10^2	10^3	10^4	10^5	10^6	10^7	10^8	10^9	10^{10}	10^{11}
Salmonella newport						+	++					
Salmonella bareilly						+	++					
Salmonella anatum (I)					-	+						
(II)					-	-	-	+				
(III)					-	-	++	+	+			
*							++	+	+			
Salmonella meleagridis (I)					-	-	-	++				
(II)							-	++				
(III)						-	+	++				
*								+++	+++			
Salmonella derby						-	-	++				
Salmonella pullorum (I)					-	-	-	-	-	-	++++	
(II)					-	-	-	-	-	++++		
(III)					-	-	-	-	-	++++		
(IV)					-	-	-	-	-	++		

TABLE 160. (continued)

ORGANISM	Challenge doses											
(strain)	10^0	10^1	10^2	10^3	10^4	10^5	10^6	10^7	10^8	10^9	10^{10}	10^{11}
Escherichia coli (0111:B4)							+++		+++	+++		
(055:B4)									+++	+++	++++	
(06:H16)									++		+++	
(0124:K72:H-)									+		(+)	
(0143:K?:H-)					(+)		(+)		+++			
(0144:K?:H-)					-		+		+++			
(0148:H28)									+		++++	
Streptococcus faecalis var. liquefaciens									-	+	++	
Clostridium perfringens type A (Heat-resistant)									++	+++		
Clostridium perfringens type A (Heat sensitive)										++++		
Endamoeba coli	(+)	(++)	(++)	-	(++++)							
Giardia lamblia	-	(++++)	(++)		(++++)	(++++)	(++++)					

TABLE 160. (continued)

ORGANISM (strain)	Challenge doses 10^0	 10^1	 10^2	 10^3	 10^4	 10^5	 10^6	 10^7	 10^8	 10^9	 10^{10}	 10^{11}
Norwalk agent (virus)1 ++ ++ 2 +++ 3 +												
Hepatitis virus A +++ (fecal ++ viltrates)												

- = 0, + = 1-25, ++ = 26-50, +++ = 51-75, ++++ = 76-100 percent of volunteers developing illness.

* Refeeding trials of volunteers who mo before became infected by the same strain.

** 1,2,3 refers to serial passages of stool filtrates.

(+) Infections without illness.

1 = cholera-like diarrhea.

outbreak of giardiasis by G. lamblia was reported in September 1976 in Idaho. The source was purported to be from untreated surface water of an individual water system (561).

Apparently, the cysts of G. lamblia survive in water and remain infective for 16 days. During 1969 to 1973, seven outbreaks involving 193 people were reported in the United States. During October 1954 to March 1955, there was a suspected waterborne outbreak of 50,000 cases of giardiasis in Portland, Oregon. The outbreak was not reported, because of the failure to isolate the organism from the suspected water source (909). Epidemic giardiasis among American travelers to the Soviet Union has been reported since 1970; the latest outbreak was reported in October 1975 (1581). Sporadic single cases or occurrences of giardiasis with recent exposure to untreated mountain or pond water have been noted (941). In an experiment where adult humans were given challenge doses of Giardia lamblia, 76 to 100 percent infected with a dose containing 10 organisms did not become ill. Similar results were observed with doses containing up to a million organisms (see Table 160).

Human schistomiasis is another parasitic infection. Although it does not occur naturally in North America, human schistomiasis is a difficult problem to control in other parts of the world. Schistosoma mansoni in Africa and S. japonicum in Japan are the principal etiological agents of schistosomal dysentery, parasitic periportal cirrhosis, and secondary portal hypertension. S. haematobium is the etiological agent of schistosomal haematuria (the passage of blood in urine due to schistosoma) and schistosomal carcinoma (cancer due to schistosoma) of the bladder. The schistosomal worms are known as blood flukes; their primary hosts are aquatic snails (1212). "Swimmer's itch," a schistosomiasis in which the primary hosts are waterfowl, is an accidental infection by the cercariae. Because man is not a natural host, their development results in local irritation and itching.

In mid-May 1976, 11 Girl Scouts noticed a transient, pruitic, erythematous macular rash on their lower extremities 15 to 20 min after wading in a lake in Shadow Cliffs Regional Park, California. Four days later, 7 of the girls developed superimposed papules. The lesions cleared after a few days. Several other individuals were similarly affected after swimming in Castaic Lake, California (183). These cases are typical of "swimmer's itch" of schistosome dermatitis, caused by a variable allergic response to schistosome cercariae whose definitive hosts are birds and small semiaquatic mammals. The short-lived, free swimming cercariae are liberated from snails (the intermediate hosts) and penetrate the human skin during water contract in infected fresh water lakes or man-made water impoundments. After skin penetration, the cercariae die.

TABLE 161. AVERAGE VALUES FOR C. ALBICANS, COLIFORM AND FECAL COLIFORM COUNTS AND TOC (TOTAL ORGANIC CARBON) DETERMINATIONS IN THE ESTUARINE WATER SAMPLES, LONG ISLAND, NEW YORK (631)

Estuarine Stations	*C. albicans* cells/ℓ	Coliform Total MPN/100 mℓ	Coliform Fecal MPN/100 mℓ	TOC mg/ℓ
Station 1	4,245	338	33	8
Station 2	4,003	298	12	8.4
Station 3	9,555	1,962	490.5	8.7

This fungus retained an infectivity and pathogenicity to mice after it was exposed to sea water for eight weeks (631). In the open ocean, there is a concentration of 200 to 300 fungal cells/ℓ; in moderately contaminated beach areas, there are levels of 10,000 to 20,000 cells/ℓ; and in heavily contaminated estuaries, up to 100,000 cells/ℓ (631). The overall trend was a gradual increase in concentrations during summer months, from June to September, after which the concentration declined (631).

Bacteria

Salmonellosis--

A wide variety of species pathogenic to man and animals belongs to the genus salmonella. Water and food, as well as personal contact, are the main routes for transmission of the species from man to man.

Among the three distinct forms of salmonellosis in man, typhoid fever - caused by *S. typhosa* - is the most severe enteric fever form, and man is the only host. Salmonella septicemia, most commonly caused by S. choleraesuis, is relatively rare. This bacteria is not particularly common in humans, but has a predilection for swine. Salmonella are most commonly encountered in acute gastroenteritis. Serotypes (types based on antibodies) in excess of 1,500 have been identified. Most, in contrast to *S. typhosa*, are not host specific.

The death rate, due to typhoid fever in the United States in 1900, was 31.3/100,000 population; however, at present, death due to this disease is practically nonexistent, as shown in Table 162 (308, 613).

TABLE 162. U.S. MORTALITY FROM SELECTED CAUSES RELATED TO WATER POLLUTION (308).

Cause of Death	Death Rate per 100,000					
	1920	1930	1940	1950	1960	1967
Typhoid & paratyphoid fever	7.6	4.8	1.1	0.1	0.0	0.0
Dysentery	4.0	2.8	1.9	0.6	0.2	0.1
Gastritis, duodenitis, enteritis, & colitis	53.7	26.0	10.3	5.1	4.4	3.8

Amoebic meningoencephalitis has been reported in different parts of the world, including the United States. The etiologic agent is Naegleria fowleri, a free-living soil amoeba. Epidemiologically, the amoebae have been aspirated by swimmers into their nasopharynx, resulting in such clinical conditions as meningoencephalitis, brain abscesses, and lung abscesses (277). The disease can be contracted by the intensive passage of amoebae from one human to another in overcrowded swimming pools. The disease is also associated with streams connected to canals that receive warm industrial effluent. In addition, reptiles that harbor such amoebae can transmit the disease.

An outbreak of ascariasis occurred after World War I; one out of every three surgical patients at a hospital in Le Havre, in the 2 yr following cessation of the war, was found to have the disease (1509). Also, following World War II, a 40 percent incidence of ascariasis was reported in Darmstad and was attributed to a widespread breakdown of sanitation practices that occurred in Germany during the latter part of the war (1509).

Ova from the giant roundworm Ascaris lumbricoides, the pinworm Oxyuris vermicularis, the whipworm Trichuris trichuria, the tapeworm Taenia saginata, and possibly the hookworm are reported to be present in wastewater (428). Heavy infestations of roundworms found in some European cities are related to the use of night soil, which is also known to be responsible for about 20 percent of the recurrent infections of amoebiasis and hookworms (459). The cestode eggs are transmitted to the intermediate host (swine or cattle), and man is infected by eating the meat of the intermediate host. This is of greatest concern when sewage sludge is spread in agricultural areas.

Man is known to be the host and reservoir of A. lumbricoides, whose ova are excreted in the feces of infected individuals. Ova of these intestinal parasites require several days before maturing to an infective stage. Under the most favorable conditions, ascarids require 10 days to mature; trichurids, 21 days; and hookworms, 6 days. Hence, these parasites are unlikely to pose health threats if the water is used for drinking, since they would be eliminated from the human body before maturation. However, use of such water for food processing, garden watering, or simple aesthetic activities may require the effective removal or destruction of the parasites (1509).

Free-living nematodes are widely found in municipal water supplies. Their potential as carriers of enterococci, salmonella, and shigella has been demonstrated (1509). Although free-living nematodes are not important as health threats in conventional waterworks practice, their significance in a reclaimed water situation needs evaluation.

Dracontiasis, or guinea worm infection, is a disease primarily associated with poverty (especially inadequate water supplies and wastewater treatment), and is common in India and West Africa. Vector species of Cyclops persisting in ponds drinking water must be controlled (usually by the use of Abate at a concentration of 1 mg/ℓ).

In moderate climates, the human contribution of ova to wastewater would appear to be no greater than 10 percent, but may reach 30 percent in subtropical regions such as the southern extremities of the United States. The remainder of the ova is of animal origin. Various authors have reported 59 to 80 worm eggs/ℓ sewage (428). The eggs are generally resistant to environmental conditions, having a thick outer covering to protect them against desication. In one study, 90 percent of ascaris ova was destroyed after 15 days at 29°C; the ova may survive for up to 60 days at 40°C (428).

In summary, it may be stated that a large quantity of a variety of ova from parasitic worms may be present in wastewater, and that the ova possess a high degree of resistance to many environmental stresses.

Fungi

Candida albicans, a pathogenic, yeastlike fungus, is an asporogenous (non-spore-forming) yeast that develops pseudohyphae (a kind of filamentous structure). Large, round, thick-walled chlamydospores - the morphological characteristic of the genus candida - are frequently, though not always, present.

C. albicans has been found in the feces and skin of several animal species other than man. The fungus rarely occurs in soil. The prevalence of candida in feces in different regions of the United States varies from 19.3 to 44.6 percent of the total population (631). Seventy percent of the inhabitants of Baghdad have the fungus in their feces. Another important infective area in the human body is the female vagina; 43 percent of the female population in the United States are carriers (631).

C. albicans may cause oral thrush, corneal ulcers, and other ocular infections. One survey analyzed 20 weekly samples collected at three stations on the North Shore of Great South Bay, Long Island. It was reported that the estuarine water samples contained between 1,000 to 11,000 or more cells/ℓ (631). These results are shown in Table 161.

Relatively few outbreaks of typhoid fever and salmonellosis associated with drinking water were reported in the United States during 1971 to 1973 (425, 910). The reported isolation rates for humans in the United States in 1972 was 12.5/100,000; the fatality rate between 1962 and 1972 was 0.43 percent, mostly among the very young and very old.

In 1974, 23,833 isolations of salmonella were reported to the Center for Disease Control, a decrease of 2,855 cases (10.7 percent) from the previous year. As in 1973, *S. typhimurium* (30.8 percent), *S. newport* (6.9 percent), and *S. enteritides* (6.0 percent) were the first, second, and third most commonly isolated serotypes, respectively (1192). The annual incidence of reported human isolations of salmonella has remained relatively constant since 1963 (1192). The seasonal incidence for the period from 1967 to 1974 shows a consistent pattern, with the greatest number of isolations reported in July through November, and the fewest reported in February through April (1192).

The ages of infected persons reveal that 66.8 percent of the 17,229 isolations was from persons less than 20 yr of age. Similarly, this age group showed the highest infection incidence for the years from 1963 through 1973 (1192).

Several serotypes (e.g., *S. weltevreden*, *S. panama*, and *S. oslo* in Hawaii; *S. newport* and *S. javiana* in the southern states) have definite regional patterns, for reasons that are not clear.

From 1962 to 1974, 143 deaths were reported in 34,291 persons involved in 499 outbreaks. This resulted in a case-fatality ratio of 0.42 percent.

In 1974, in 18 of the 34 outbreaks of salmonellosis (involving at least 4,011 persons), 5 outbreaks were caused by contaminated poultry, 5 by beef or beef products, and 3 by dairy products. In 6 of these outbreaks, person-to-person transmission was thought to be responsible; none was reported to be transmitted by water. In 1973, 3 typhoid outbreaks (totalling 217 cases) and 1 outbreak of salmonellosis (3 cases) occured in semipublic or individual water supplies. In 1974, an outbreak involving several hundred people on a cruise was reported; epidemiologic investigation failed to clearly implicate either food or water. Three epidemiologic investigations of turtle-associated salmonellosis were also found in the literature (270).

According to experimental data, the dose required to bring about human cases of typhoid fever is surprisingly high (1576). Human volunteers were challenged with one dose of a given number of *Salmonella typhosa*. The results are summarized in Table 163.

TABLE 163. RELATION OF DOSAGE OF *S. TYPHOSA* TO DISEASE (1576)

Number of Viable Cells *S. typhosa*	Total Volunteers Challenged	Number with Disease
10^9	42	40 (95%)
10^8	9	8 (89%)
10^7	32	16 (50%)
10^5	116	32 (28%)
10^3	14	0 (---)

With the salmonella species isolated from spray-dried eggs, human volunteers were challenged orally; the results are summarized in Table 164. Existence of varying degrees of virulence among the species and strains are shown. Similar results are shown in Table 160.

TABLE 164. DOSE OF VARIOUS SPECIES AND STRAINS OF SALMONELLA THAT CAUSED DISEASE IN HUMAN VOLUNTEERS (1571)

Salmonella Species/Strain	Dose at Which 50% or More Develop Clinical Disease
S. meleagridis I	50,000,000
S. meleagridis II	41,000,000
S. meleagridis III	>10,000,000
S. anatum I	860,000
S. anatum II	67,000,000
S. anatum III	4,700,000
S. newport	1,350,000
S. derby	15,000,000
S. bareilly	1,700,000
S. pullorum I	>1,795,000,000
S. pullorum II	>163,000,000
S. pullorum III	>1,295,000,000
S. pullorum IV	1,280,000,000

Shigellosis

Shigella cause bacillary dysentery in man and in higher apes. Although person-to-person transmission is the predominant mode of spreading shigellosis, waterborne outbreaks have played a significant role in the overall epidemiology of the disease in the United States (1320, 1582). In 1975, 14,757 shigella isolations from humans were reported to the Center for Disease Control (CDC). This was a decrease of 24.0 percent from the 19,420 isolations reported in 1974 (1249). Utilizing population estimates for July 1, 1975, approximately 69.2 isolations were reported for each million population of the United States in 1975. Shigella sonnei (60.3 percent) was the most common etiological agent followed by S. flexneri (38.2 percent). Between January 17 and March 15, 1974, approximately 1,200 cases of acute gastrointestinal illness occurred in Richmond Heights, Florida. The outbreak was caused by a failure in the chlorination process of well water, which allowed insufficiently chlorinated water from a contaminated well (located near a church's septic tank) to be distributed to the community (270).

Most instances of Shigella-induced illnesses reported in the past several years have involved small wells, temporary breakdowns of chlorination systems in water supply, and swimming in waters contaminated with sewage. The isolation rate in the United States is approximately 15/1,000,000 population. Up to 25 percent of adult humans may be infected and show clinical response to *Shigella dysenteriae* doses of 10 cells; 25 to 50 percent, to doses of 100 cells; and up to 100 percent, to doses of 10^4 cells (see Table 160). These results indicate that it would take smaller doses to show clinical response for shigella than it would take to show response for salmonella.

Cholera

Cholera, which is fully controlled in the United States, has created a major global public health problem in India, Italy, Portugal, and many other countries. In the United States, only one case (in the Gulf Coast town of Port Lavaca, Texas) has been reported since 1911. Transmission was associated with a well that was contaminated by leachate from a septic tank system.

The etiological agent of cholera is *Vibrio cholerae*, Biotype El Tor, Serotype Inaba or Ogawa. Man is the only known natural host, and, since a prolonged carrier state is uncommon, the disease must be maintained by an unbroken chain of mild, subtle infections. Cholera is a serious disease, similar to typhoid fever but more rapid in onset, more virulent, and more often fatal. Death rates of 25 to 85 percent are commonly reported (1571).

The primary means of cholera transmission is the drinking of contaminated water. Also associated with cholera is the eating of fish caught in contaminated waters. In the event that imported cases occurred in the United States, it is felt that the risk of spread would be minimal because of modern, industrialized sanitary engineering as well as responsible medical therapy. It is also reported that vaccination is not needed to control imported cases or outbreaks that may occur in the United States (270).

The minimum doses of vibrio that cause clinical symptoms in adult humans may vary (depending upon the strains of vibrio that are used) from 10^3 cells (in up to 50 percent of the individuals) to 10^5 cells (in up to 75 percent of the individuals), as shown in Table 160.

Gastroenteritis

There are many reports of waterborne gastroenteritis of unknown etiology in which bacterial infections are suspected. These include outbreaks characterized by nausea, vomiting, diarrhea, and fever, for which no specific etiologic agent could be identified.

An epidemiological study of the impact of wastewater pollution on marine bathing beaches was conducted during the summers of 1973 and 1974 at the Coney Island and Rockaways beaches in New York (179). A statistically significant finding was observed: the rate of gastrointestinal symptoms among swimmers compared to nonswimmers was higher at Coney Island, where the densities of the indicative organisms such as E. coli and fecal streptococci were significantly higher than the densities at Rockaways Beach (Tables 165 and 166). It was concluded that there were measurable health effects associated with sewage polluted waters (179).

In 1971, a waterborne gastroenteritis outbreak was reported in Pico Rivera, California, in which 11,000 residents became ill with diarrhea and abdominal cramps. No pathogens were isolated from any cases. The source of water was responsible for the outbreak; chlorination at the reservoir had been interrupted when the chlorine supply was exhausted. One of the major outbreaks involving over 1,000 persons occurred at Crater Lake National Park, Oregon, in July 1975. The illness was reported to be associated with sewage-contaminated water (945). Enterotoxigenic E. coli, Serotype 06:H16, was isolated from ill park residents and from the park's water supply (426).

It was noted that the attack rate of acute "summer diarrhea" on the Fort Apache Reservation, Arizona, during 1971, rose simultaneously with rainfall, temperature, and bacterial contamination of water sources (1520). Nonenteropathogenic E. coli capable of producing enterotoxin was isolated.

In the study of clinical response of adult humans to challenge doses of E. coli, it has been demonstrated that very large doses of cells (about 10^8) would be necessary to show clinical response in 75 percent of the individuals tested (Table 161). With the E. coli strain of 0111:84, doses of only 10^6 cells were necessary for a similar response. However, it should be noted that the digestive tract in most individuals is populated by normal flora, of which E. coli is the most abundant and most characteristic; about 10^9/g of feces is common (85).

TABLE 165. MEAN INDICATOR DENSITIES AT THE CONEY ISLAND AND ROCKAWAYS BEACHES, NEW YORK, DURING 1973 AND 1974 TRIALS (179)

	Log Mean Recovery/100 mℓ			
	1973		1974	
Indicator	Coney Isl.	Rockaways	Coney Isl.	Rockaways
Total coliforms	983*	39.8	1213*	43.2
Fecal coliforms	165*	21.5	565*	28.4
Escherichia coli	174*	24.8	15.3*	2.4
Klebsiella	112*	13.7	59.2*	3.5
Enterobacter-citrobacter	530*	11.1	434	6.6
Fecal streptococci	91.2	21.8	16.4*	3.5
Pseudomonas aeruginosa	30.4	6.5	45.8*	3.1
Aeromonas hydrophila	25.3	26.5	9.6	4.9
Vibrio parahaemolyticus	ND	ND	54.5	32.8

*Significantly different at 95 percent confidence level.

TABLE 166. SYMPTOM RATES IN PERCENT AT CONEY ISLAND AND ROCKAWAYS BEACHES, NEW YORK, DURING 1973 AND 1974 TRIALS (179)

Symptom Group	Rates in Percent for Symptom Groups in											
	1973						1974					
	Coney Island			Rockaways			Coney Island			Rockaways		
	S	NS	×	S	NS	×	S	NS	×	S	NS	×
N	474	167		484	197		1961	1185		2767	2156	
Resp.	12.9	10.2	2.7	18.0[ab]	11.7	6.3	7.2	6.4	0.8	8.3	7.8	0.5
G.I.	7.2[a]	2.4	4.8	8.1	4.6	3.5	4.2[a]	2.6	1.6	3.9	3.5	0.4
Other	9.9	6.6	3.3	9.1	8.6	0.5	7.3	6.7	0.6	8.6	7.7	0.9
"Severe"	5.9	4.2	1.7	6.0	5.6	0.4	3.8	2.9	0.9	3.0	2.6	-0.4

[a] Significantly (P = 0.5) higher than nonswimmers.

[b] Significantly (P = 0.5) higher than other beach.

N-sample size; S-swimmers; NS-nonswimmers; ×-difference

GI - gastrointestinal; Resp. - respiratory

Leptospirosis

Etiological agents of leptospirosis (an infection caused by leptospira) constitute approximately 150 different serotypes categorized on the basis of their agglutinogenic properties (1577). The pathogenic serotypes are otherwise indistinguishable by morphology or biochemical activity.

Recognition of two species, Leptospira interrogans and L. biflexa, has been proposed for the so-called pathogenic and saprophytic leptospires, respectively (1577).

Generally, leptospirosis, typically a disease of animals, has been regarded as an occupational disease that occurs primarily in workers associated with wastewater, rice, sugar cane, farms, and slaughter yards. The disease is transmitted to man by direct contact or via water contaminated by urine from infected wild and domestic animals. Leptospirosis is endemic in some parts of the world. However, 11 outbreaks in nonendemic areas during 1939 to 1959 were associated with swimming or wading in contaminated water (270, 1577), indicating the potential for contacting the disease as a result of recreational activity.

At the same time, leptospirosis has become a more recognizable human health problem in recent years, because of better identification of symptoms and improved methods of diagnosis. In rural populations in France, the percentage of infection by source was as follows: water or mud, 21 percent; animals, 53 percent; and water and animals, 25 percent. In nonrural populations, however, 80 percent of the infections was acquired by contact with contaminated water (1572). It was also noted that subclinical leptospirosis in humans was not infrequent and may be a significant public health hazard (1572). Information on the infective dose of leptospira to humans is not available from the literature surveyed. Leptospira can apparently penetrate intact skin, assisted by cuts, abrasions, and immersion. Incidences of infection from swimming also suggested penetration via the mucous membranes of the mouth or nasopharynx (270). The nesting site of leptospires in natural hosts is the lumen of nephritic tubules, from which they are shed into the urine (1577). Pathogenic leptospires can survive for three or more months in neutral or slightly alkaline waters, but do not persist in brackish or acidic water (1577). In 1975, two outbreaks of leptospirosis were attributed to swimming in contaminated surface water. Seven children in Tennessee developed infection with L. interrogans serotype grippotyphosa after swimming in a small local stream. Two persons in Louisiana became infected with leptospires of the serotype icterohaemorrhagiae after bathing in a man-made lake (426).

Tuberculosis

The possibility that tuberculosis may be transmitted by sewage has frequently been considered in connection with the disposal of wastewaters from hospitals, tuberculosis sanitaria, dairies, and slaughter houses; the possibility has even been considered in connection with domestic sewage in general. Concern has been expressed about the danger of human and animal infection, particularly where these waters are reused (509).

The presence of mycobacteria in wastewater has been extensively studied since around 1900 - the time of the first findings of the bacteria in feces (428). The recovery of *Mycobacterium tuberculosis* (the bacteria that cause tuberculosis) is difficult, even from favorable sources such as sputum; recovery from sewage is much more difficult because of the presence of other bacteria.

The tubercle bacilli are present in the sputum and feces of tuberculosis patients. The wastes from institutions that treat the patients will almost always contain large numbers (4×10^5 to $10^7/\ell$) of tubercle bacilli (509). Significant numbers (about $3/\ell$ in the effluent of a plant producing about 7,600 gpd of milk) of virulent tubercle bacilli are also found in the wastes from some dairies. A cow suffering with tuberculosis of the udders discharges about 1.5×10^8 tubercle bacilli/day (509). *M. balnei*, which causes granuloma, may be present in chlorinated water used for swimming pools (580). Much of the data on survival of tubercle bacilli in sewage indicate that, under laboratory conditions. the bacteria can be infective for 6 months in sewage and for up to 24 months in feces (509).

Contaminated water can produce typical tuberculosis in humans in some instances. The first clear-cut cases of human infection were reported in 1947. From 1947 to 1953, nine cases of tuberculosis were described in humans who aspirated polluted water into their lungs after swimming and nearly drowning in contaminated water (509).

There were not much data available in the literature on the infectious dose to humans of tubercle bacilli in wastewater. In an experiment with guinea pigs, it was reported that 80 percent of the animals contracted the disease when they were fed with grass that had been sprayed with more than 4×10^6 tubercle bacilli. Calves that had been similarly fed also succumbed (509).

The majority of studies carried out on mycobacteria has focused on the presence of *M. tuberculosis* in sanitarium wastes. These studies may not provide a realistic picture of the danger of infection from contaminated water.

The potential health hazards of M. tuberculosis being transmitted through sanitarium wastewaters, to drinking supplies, and back to man in general appears to be quite remote.

Other Sources of Bacterial Infection

Vibrio parahaemolyticus is a marine bacteria that is known to be associated with common-source gastroenteritis outbreaks when contaminated seafoods are consumed. The bacteria is not directly involved with wastewater; however, its frequent occurrence in the coastal waters and aquaculture farms alarms consumers of seafoods as well as bathers in contaminated waters (270). Between 1969 and 1972, 13 common-source outbreaks were recognized in the United States. This infectious agent has been the leading cause of food poisoning in Japan for some years.

Klebsiella pneumoniae is one of the coliform group members, closely resembling Enterobacter aerogenes. In one case, it was reported that K. pneumoniae were isolated from pulp and paper wastewater effluents and represented as much as 80 percent of the total coliform bacteria present (722). Although the origin of the bacteria in the pulp mill wastewaters was not resolved, fecal origin was a strong possibility. Whether ingested K. pneumoniae are pathogenic and a hazard to human health should be determined. They are a respiratory pathogen.

Pseudomonas aeruginosa has been an etiological agent for many types of infections in hospitals. Contaminated sinks and flower vases are some of the well-known sources of P. aeruginosa (270, 1572). The microbiological characteristics of sewage discharged by hospitals are interesting when compared with those of residential wastewaters, because counts of all the organisms tested, except P. aeruginosa, were lower in the hospital effluent (493).

Clostridium perfringens is known to cause gas gangrene and is found in anaerobic environments such as sludge, mud, and sediment in aquatic and marine environments. It has been shown that a dose of at least 10^9 cells of the bacteria would be necessary to demonstrate clinical response in 75 to 100 percent of the individuals tested (see Table 160).

Viruses

In the past, transmission of waterborne viral diseases was rarely recognized, due largely to lack of sensitive virus-detection methods and precise quantification. With improved techniques for concentrating viruses from large water samples, increasing occurrences of viruses in water and wastewaters have been reported. Viral transmission through water may take place in various ways: bathing in contaminated water, eating contaminated seafoods, drinking from untreated or improperly treated water sources, or contacting contaminated waters. Enteric viruses have been investigated with greater emphasis than any other group of viruses, mainly because any virus excreted in the feces and capable of producing infection when ingested is theoretically transmissible by water. The human enteric viruses and the diseases associated with them are listed in Tables 167 and 168.

TABLE 167. THE HUMAN ENTERIC VIRUSES AND THE DISEASES ASSOCIATED WITH THEM (1571)

Virus Subgroup	No. of Types	Disease
Polio virus	3	paralytic poliomyelitis, aseptic meningitis
Coxsackie virus		
Group A	26	herpangina, aseptic meningitis, paralysis pleurodynia,
Group B	6	aseptic meningitis, acute infantile myocarditis
ECHO virus	34	aseptic meningitis, rash and fever, diarrheal disease, respiratory illnesses
Infectious hepatitis	1(?)	infectious hepatitis
Reovirus	3	fever, respiratory infections, diarrhea
Adenovirus	32	respiratory and eye infections

Infectious Hepatitis

Apart from theoretical considerations, there are very few viruses for which epidemiological evidence suggests transmission by water. Infectious hepatitis (hepatitis A) is the only disease caused by an agent having the characteristics of a virus for which evidence of waterborne transmission has been accepted by all workers in the field (976). Therefore, it is regarded as the viral disease of greatest importance in wastewater.

In 1973 alone, a total of 59,200 cases of viral hepatitis A, B, and a type unspecified were reported (426). In 1974, a total of 59,340 cases of viral hepatitis - hepatitis A, B, and type unspecified - were reported to CDC. This represents a rate of 28.1 cases/100,000 population, approximately the same rate as for 1973. Since 1971, 1974 is the first year to have shown rate

TABLE 168. THE HUMAN ENTERIC VIRUSES THAT CAN BE WATERBORNE AND KNOWN DISEASES ASSOCIATED WITH THESE VIRUSES (1366)

Group	Subgroup	No. of Types or Subtypes	Disease Entities Associated with These Viruses	Pathologic Changes in Patients	Organs Where Virus Multiplies
Enterovirus	Poliovirus	3	Muscular paralysis	Destruction of motor neurons	Intestinal mucosa, spinal cord, brain stem
			Aseptic meningitis	Inflammation of meninges from virus	Meninges
			Febrile episode	Viremia and viral multiplication	Intestinal mucosa and lymph
	Echo virus	34	Aseptic meningitis	Inflammation of meninges	Meninges
			Muscular paralysis	Destruction of motor neurons	Intestinal mucosa, spinal cord, brain steam
			Guillain-Barre's Syndrome	Destruction of motor neurons	Spinal cord
			Exanthem	Dilation and rupture of blood vessels	Skin
			Respiratory diseases	Viral invasion of parenchymiatous of respiratory tracts and secondary inflammatory responses	Respiratory tracts and lungs
			Diarrhea		
			Epidemic myalgia	Not well known	
			Pericarditis and myocarditis	Viral invasion of cells with secondary responses	Pericardial and myocardial tissue
			Hepatitis	Viral invasion of liver cells	Liver parenchyma
	Coxackie virus	>24	Herpangina	Viral invasion of mucosa with secondary inflammatory responses	Mouth
			Acute lymphatic pharyngitis		Lymph nodes and pharynx

TABLE 168. (continued)

Group	Subgroup	No. of Types or Subtypes	Disease Entities Associated with These Viruses	Pathologic Changes in Patients	Organs Where Virus Multiplies
Enterovirus (continued)	Coxsackie virus A		Aseptic meningitis	Inflammation of meninges	Meninges
			Muscular paralysis	Destruction of motor neurons	Intestinal mucosa, spinal cord, brain stem
			Hand-foot-mouth disease	Viral invasion of cells of skin of hands and feet and mucosa of mouth	Skin of hands and feet and much of mouth
			Respiratory disease	Viral invasion of parenchymiatous of respiratory tracts and secondary inflammatory responses	Respiratory tracts and lungs
			Infantile diarrhea	Viral invasion of cells of mucosa	Intestinal mucosa
			Hepatitis	Viral invasion of liver cells	Parenchyma cells of liver
	Coxsackie virus B	6	Preicarditis and myocarditis	Viral invasion of cells with secondary responses	Pericardial and myocardial tissue
			Pleurodynia	Viral invasion of muscle cells	Intercostal muscles
			Aseptic meningitis	Inflammation of meninges	Meninges
			Muscular paralysis	Destruction of motor neurons	Intestinal mucosa, spinal cord, brain stem
			Meningoencephalitis	Viral invasion of cells	Meninges and brain
			Pericarditis, endocarditis, mycarditis	Viral invasion of cells with secondary responses	Pericardial and myocardial tissue
			Respiratory diseases	Viral invasion of parenchymiatous of respiratory tracts and secondary inflammatory responses	

TABLE 168. (continued)

Group	Subgroup	No. of Types or Subtypes	Disease Entities Associated with These Viruses	Pathologic Changes in Patients	Organs Where Virus Multiplies
Enterovirus (continued)	Coxsackie Virus B (continued)		Hepatitis or rash	Viral invasion of liver cells	Liver parenchyma
			Spontaneous abortion	Viral invasion of vascular cells (?)	Placenta
			Insulin-dependent diabetes	Viral invasion of insulin producing cells	Langerhans cells of pancreases
			Congenital heart anomalies	Viral invasion of muscle cells	Developing heart
Reovirus		6	Not well known	Not well known	
Adenovirus		31	Respiratory diseases	Viral invasion of parenchymiatous of respiratory tracts and secondary inflammatory responses	Respiratory tracts and lungs
			Acute conjunctivitis	Viral invasion of cells and secondary inflammatory responses	Conjunctival cells and blood vessels
			Acute appendicitis	Viral invasion of mucosa cells	Appendia and lymph nodes
			Intussusception	Viral invasion of lymph nodes (?)	Intestinal lymph nodes (?)
			Sub acute thyroiditis	Viral invasion of parenchyma cells	Thyroid
			Sarcoma in hamsters	Transformation of cells	Muscle cells
Hepatitis		>2	Infectious hepatitis	Invasion of parenchyma cells	Liver
			Serum hepatitis	Invasion of parenchyma cells	Liver
			Down's Syndrome	Invasion of cells	Frontal lobe of brain, muscle, bones

increases for two quarters; the increase in cases began in the fourth quarter of 1973. The seasonal variation noted in the 1950s and early 1960s was not seen in 1974. The 48,709 cases of acute hepatitis A and hepatitis, type unspecified, constituted 82.1 percent of the total cases of viral hepatitis reported in 1974 (562). Waterborne outbreaks of hepatitis A continue to occur in the United States. From 1971 to 1973, these documented outbreaks were associated with contaminated drinking water from either municipal, semipublic, or individual water systems (270, 910). Use of contaminated spring or groundwater without proper treatment or disinfection, and back-siphonage of contaminated water into the distribution system were reported to be the causes of the outbreaks. The majority of documented hepatitis A outbreaks in municipal water systems in the United States between 1946 and 1971 occurred as a result of distribution system contamination, primarily through cross connections and back-siphonage.

Two outbreaks of shellfish-associated hepatitis involving 285 cases were reported in 1973 (426). Both outbreaks - one in Georgia and the other in Texas - were associated with the eating of raw oysters. Epidemiologic evidence suggested that two particular bays contaminated by flooding were the source of the contaminated oysters.

A recent hepatitis outbreak of 14 cases was reported to be associated with swimming in a grossly contaminated lake in North Carolina, and with ingesting water from that lake. This is the first time that such a definite case has been made for the potential of contracting this disease while swimming in sewage-polluted water. The probable fecal-oral transmission of infectious hepatitis made the waterborne route possible. This mode of transmission was vividly illustrated by several large epidemics that took place in 1955 to 1956, especially in India where 28,745 cases occurred (976).

As far as the magnitude of waterborne infectious hepatitis is concerned, the water route still only accounts for up to 1 percent of reported cases at any time for which information is available (976).

Despite the increased interest and concern in infectious hepatitis, its infectious agent has not yet been isolated and cultured. One recent report (404) using microscopic techniques was able to show the presence of viruslike particles, immunologically distinct from hepatitis B, in infected stools.

Poliomyelitis

The infectivity of feces from persons with poliomyelitis and the characteristic fecal excretion of the diseased persons have been documented for years. The polio virus has been sought and detected in sewage. Accordingly, the water route of transmission has been implicated in several outbreaks of poliomyelitis (see Table 169). Many cases of epidemics of poliomyelitis were attributed to waterborne transmission through contaminated or untreated water, but the evidence is not sufficient. It appears that water transmission of the polio virus may be a rare occurrence in the United States, but common in parts of the world lacking adequate sanitary facilities. Six of the outbreaks attributed to drinking water occurred in Sweden during the 1930s and 1940s (976) and led to the early recognition of the importance of the fecal-oral route in poliomyelitis.

Viral Gastroenteritis

When a recognized pathogen cannot be isolated in cases of gastroenteritis and diarrhea, the term viral is often used to describe the symptoms. It is quite possible that forms of gastroenteritis and diarrhea transmissible from person to person are due to viruses.

Gastroenteritis and diarrheal disease are believed to have accounted for approximately 60 percent of all epidemics of waterborne diseases throughout history. The number of these cases that was due to viral agents is not known; if, however, only a small portion of the cases was due to viral agents, this number would still be quite substantial.

A virus-like particle, similar in appearance but immunologically distinct from the hepatitis A, has been reported to be associated with an acute infectious nonbacterial gastroenteritis (680). Shellfish-associated gastroenteritis has also been reported (332).

Other Viral Diseases

Outbreaks of pharyngo-conjunctivitis, caused by adenovirus are reported in association with swimming in contaminated pools (1568). The implication of waterborne transmission of the disease was based on epidemiologic evidence; no viruses were obtained from the water.

Successful isolation of the causative virus demonstrated the water transmission of coxsackie virus type A16 to children who swam in lake water with relatively high fecal coliform counts.

Infective Dose of Viruses

Virologists feel that one plaque-forming unit (pfu) - one viral particle that grows in the laboratory media - constitutes an infectious viral dose. However, there is a difference between infection and disease (1366): the diseased person manifests

TABLE 169. PUBLISHED REPORTS OF POLIOMYELITIS ATTRIBUTED TO CONTAMINATED DRINKING WATER (976)

No.	Yr of occurrence	References	Country	Place or type of population	No. cases attributed to supply
1	1913(?)-39	Spaak, 1941	Sweden	Rural district	10[b]
2	1944	Kling, 1947	Sweden	Town	63
3	1944	Kling, 1947	Sweden	Town	53
4	1948	Faahraeus et al., 1950	Sweden	Stockholm suburb	9
5	1948	Faahraeus et al., 1950	Sweden	Town	63[c]
6	1949	Huss et al., 1952	Sweden	Malmoe	138
7	1952	Bancroft et al., 1957	U.S.A.	"Huskerville"	45
8	1953	Little, 1954	Canada	Edmonton	75[b]

[a] Unadjusted rate among persons presumably consuming contaminated supply.

[b] Estimated from author's data.

[c] Includes cases attributed to other modes of transmission.

TABLE 169. (continued)

No.	Attack rate per 100[a]	Character of Episode	Duration of waterborne phase	Type of supply held responsible
1	--	Sporadic cases	Yrs	Private well, pond
2	0.5	Epidemic	5 mo	Municipal system Filtered surface water
3	0.2	Epidemic	3 mo	Municipal system Untreated deep well water
4	--	Sporadic cases	3 mo	Municipal system Proximate contamination
5	0.2	Epidemic	7 mo	Municipal system Untreated deep well water
6	0.1	Epidemic	6 mo	Municipal system Filtered surface water
7	6.7	Epidemic	5 weeks	Municipal system Proximate contamination
8	<0.1[b]	Epidemic	2 weeks	Municipal system Chlorinated surface water

[a] Unadjusted rate among persons presumably consuming contaminated supply.

[b] Estimated from author's data.

[c] Includes cases attributed to other modes of transmission.

a variety of symptoms and readily recognizes that he is sick; an infected person has the material in his system, but does not necessarily show symptoms of the disease. It has been estimated that of every 100 to 1,000 people who are infected, only one will manifest the clinical symptoms of disease. However, it is not quite as simple as these statistics suggest, for an infected individual can serve as a carrier or source within the community and transmit this disease to other people (1366).

Table 170 shows the minimal infective doses of attenuated polio virus for human hosts by oral routes. The infected rate with 20 or more pfu was 100 percent; with 2 pfu, 67 percent of the individuals was infected.

TABLE 170. MINIMAL INFECTIVE DOSES OF ATTENUATED POLIO VIRUSES FOR HUMAN HOSTS BY ORAL ROUTES (1366)

Subject	Virus	Dose (pfu)	Carrier* Rate	Infected Percent
Adults	Polio virus Type I (SM Strain)	200	4/4	100
		20	4/4	100
		2	2/3	67
		0.2	0/2	0
Premature Infants	Polio virus Type III (Fox Strain)	10	2/3	67
		2.5	3/9	33
		1	3/10	30

* Number of persons developed into carrier for the virus/number of persons who had taken the virus orally.

It has been mentioned that infection with small amounts of virus in water would probably immunize individuals rather than produce disease. This may be substantiated by the fact that sewage workers continually exposed to small amounts of infected material had the lowest rate of absenteeism among all the occupation groups studied (98).

An adequate biological indicator for viruses in various waters is not currently available; however, efforts are being made to find better indicators. For example, ratios of coliphages to human enteric viruses and the coliform-virus ratio have been investigated (734, 1574). Also, a high degree of coliforms and coliphage occurring in water samples (693), as well as a yeast and two acid-fast bacilli recovered from wastewater, resist chlorination at a level sufficient to inactivate viruses. This suggests that they may be useful indicators of wastewater chlorination efficiency (383).

Infection From Microbial Aerosols

An investigation was made of the morbidity risk factors from spray irrigation with treated wastewater (1320). It was pointed out that no instances of disease have been reported from spray irrigation with chlorinated secondary effluent. This is not surprising. If aerosolized pathogens constituted a public health risk from drift and spray of aerosolized particles generated from sewage, the clinical risk would be greatest where the concentration is greatest -- in the sewage works, near the aerobic treatment unit. No evidence has been found that either long-term or newly employed operators have diseases attributed to aerosol ingestion. Indeed, some experts have suggested that the morbidity rate of sewage plant operators is less than the morbidity rate of the general public. Perhaps a more sensitive indicator would be the health of adolescent school children visiting a sewage plant for a science course field trip. A letter from the National Science Teachers Association told us that "from inquiries to the U.S. Public Health Service and the Montgomery County (Maryland) Health Department, we have learned of no reported illnesses to students following visits to sewage treatment plants, nor could we find that any guidelines have been developed or are in practice as precautions for such visits. Also, the student tours director for the Washington Suburban Sanitary Commission told us that no special precautions have been observed in their tour program, nor did he know of any illnesses to students following tours of that facility" (1320).

Many factors influence the survival of bacterial and viral pathogens in the air. Aerosols remain viable and travel greater distances with increased wind velocity, increased relative humidity, lower temperatures, and darkness. Some bacteria such as E. coli have been shown to have an extremely short life-span in aerosol form (545, 1099, 1302, 1303, 1304).

Specific pathogenic bacteria of klebsiella form a large capsule that apparently protects the organism from desication in flight. All species of this genus are known pathogens of the respiratory tract (1304).

Another pathogen of the respiratory tract, mycobacterium species, has been known to cause infections associated with contaminated water but not with aerosols (509).

A direct means of human infection by biological aerosols is inhalation. The infectivity of an aerosol is further dependent on the depth of respiratory penetration (1304). Biological aerosols in the 2 to 5μ size range are primarily captured in the upper respiratory tract. These particles are removed by bronchial cilia and may pass into the digestive tract through the pharynx (2). If gastrointestinal pathogens are present in these aerosols, a certain degree of infection may result. However, a much higher incidence of infection results when respiratory pathogens are inhaled into the alvioli of the lung.

The greatest alveolar deposition occurs in the 1- to 2-μ. range and then decreases to a minimum at approximately 0.25μ. Below 0.25μ, alveolar deposition again increases due to Brownian movement. Approximately 82 percent of 1.0 particles, 28 percent of 0.1 to 0.3μ particles, and 51 percent of 0.03μ particles are deposited in the broncheoles or the alveoli. This aspect of aerosol deposition becomes significant when viral size particles (about 0.01 to 0.1μ) are considered. Attention must therefore be paid to viral deposition, with the possible resultant infection (1304).

A summary of virus infections by the respiratory route demonstrates that extremely low doses of virus can cause infection in man (161, 1366). The tissue culture infective doses ($TCID_{50}$) cited ranged from 1.0 to 790.0 pfu. The number of infectious virus particles necessary to initiate an infection in man, as determined by antibody production or a disease state, has been reported by Sorber et al. (1304). The study utilized human volunteers who had no detectable antibody levels against the test viruses. It was observed that 28 $TCID_{50}$ of coxsackie virus A-21 in small particle aerosols (0.3 to 2.5μ) were required for a 50 percent human infective dose (HID_{50}). Only one $TCID_{50}$ of adenovirus Type IV was required for a HID_{50} when generated in small particle aerosols. Both of these enteric viruses required a higher $TCID_{50}$ to give a respiratory tract infection than did Reo virus strain NIH 1734, which required only 0.68 $TCID_{50}$. The study emphasized the importance of droplet size upon the nature of the clinical and dose response.

It is reasonable to postulate that, if disinfection of sewage is not complete and the pathogenic organisms are aerosolized, even very low numbers of these organisms may be a public health hazard (1304). Unfortunately, there are essentially no quantitative date that can be used to evaluate the risk at the present time.

SECTION III

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,REPORT BIBLIOGRAPHY

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SECTION IV

CURRENT RESEARCH

INTRODUCTION

In this section are listed research projects currently being conducted, the results of which are not included in the report text. Project titles, principal investigators, and research locations are listed under relevant headings. This information was primarily derived from an NTIS computer search and an extensive mail survey (see Section II, Introduction) conducted in November of 1975.

EXISTING LITERATURE AND RESEARCH PRESENTLY UNDERWAY

PRIMARY TREATMENT

Research programs currently being conducted are concerned primarily with dissolved air flotation and microstraining. This work is being conducted by the following organizations:

Rex Chainbelt, Incorporated
Technical Center
Treatment of Combined Sewer Overflows Using New
  Techniques of Screening and Air Flotation
W. J. Katz (Principal Investigator)
Milwaukee, Wisconsin

FMC Corporation
Environmental Equipment Division
Study of Activated Sludge Separation by Dynamic Straining
A. H. Strom and J. E. Dumanowski (Principal Investigators)
Itasca, Illinois

Philadelphia Water Department
Microstraining Pilot Tests of Combined Sewers
C. F. Guarino and G. W. Carpenter (Principal Investigators)
Philadelphia, Pennsylvania

Metropolitan Sanitary District of Greater Chicago
Performance Analysis of 15 MGD Microstrainer for Tertiary
  Treatment
C. Lue~~h~~ing (Principal Investigator)
Chicago, Illinois

Crane Company
Cochrane Division
Microstraining and Disinfection of Combined Sewer Overflows
W. A. Keilbaugh (Principal Investigator)
Norristown, Pennsylvania

Fram Corporation
Feasibility Investigation of a Self-Cleaning Strainer and Filter
S. S. Blecharczyk (Principal Investigator)
Providence, Rhode Island

American Process Equipment Corporation
Fabrication and Evaluation of an Ultrasonic Microstaining System for Treating Sewage
R. Robert (Principal Investigator)
Hawthorne, California

Cornell, Howland, Hayes and Merryweather
Fine-Mesh Screening for Primary Treatment of Storm Water Overflow
Hernandez (Primary Investigator)
Corvallis, Oregon

SECONDARY TREATMENT: ACTIVATED SLUDGE PROCESS

Research programs currently being conducted concerning the activated sludge processes are largely directed at the utilization of the activated sludge process in tandem with tertial processes. Current work is being conducted by the following organizations:

U.S. Environmental Protection Agency
National Environmental Research Center
Nitrogen Removal by the Three Stage Activated Sludge System and by a Single Stage Activated Sludge Process
D. F. Bishop (Principal Investigator)
Cincinnati, Ohio

University of Pittsburgh
Department of Civil Engineering
Aqueous Heavy Metal Removals by Biological Slimes
R. D. Neufeld (Principal Investigator)
Pittsburgh, Pennsylvania

Oklahoma State University
School of Civil Engineering
Metabolism of Components of Extended Aeration Activated Sludge
A. F. Gaudy and P. Y. Yang (Principal Investigators)
Stillwater, Oklahoma

University of New Hampshire
Department of Chemistry
Fate of Metal Ions During Treatment of Water Containing Organics
J. H. Weber (Principal Investigator)
Durham, New Hampshire

Dallas City Water Utilities Department
Removal of Heavy Metals by Activated Sludge Processes
H. W. Wolf (Principal Investigator)
Dallas, Texas

Oklahoma State University
School of Civil Engineering
Biological Treatment of a Wastewater Containing Phenol
J. H. Sherrard (Principal Investigator)
Stillwater, Oklahoma

Southwest Research Institute
Department of Chemical Engineering
Health Implications of Activated Sludge Systems
D. E. Johnson and J. T. Goodwin (Principal Investigators)
San Antonio, Texas

University of Texas
Department of Civil Engineering
Application of Oxygen to Treat Waste from Military Field Installations
J. F. Malina (Principal Investigator)
Austin, Texas

U.S. Environmental Protection Agency
National Environmental Research Center
Identification of Organic and Inorganic Compounds in Activated Sludge Treated Wastes
A. W. Garrison (Principal Investigator)
Corvallis, Oregon

Oklahoma State University
School of Engineering
Response of Completely Mixed Activated Sludge Systems to Changes in the Environment
A. F. Gaudy (Principal Investigator)
Stillwater, Oklahoma

U.S. Army
Waterways Experiment Station
Effects of Intermittent Loading on an Activated Sludge Treatment System
A. J. Green and J. L. Maloch (Principal Investigators)
Vicksburg, Mississippi

U.S. Environmental Protection Agency
National Environmental Research Center
Chlorinated Organics in Chlorine Waste Activated Sludge
A. W. Garrison (Principal Investigator)
Corvallis, Oregon

U.S. Department of Agriculture
Eastern Utilization Research Division
Recovery of Proteins from Lime-Sulfide Unhairing Effluents
W. F. Happich (Principal Investigator)
Wyndmoor, Pennsylvania

University of Texas
Department of Civil Engineering
Virus Removal in the Conventional Activated Sludge Process
J. F. Malina and K. R. Ranganathan (Principal Investigators)
Austin, Texas

FMC Corporation
Environmental Equipment Division
Activated Sludge Separation by Dynamic Straining
A. H. Strom and J. E. Dumanowski (Principal Investigators)
Itasca, Illinois

Rexnord, Incorporated
Industrial Waste Treatment by Activated Sludge
R. W. Agnew (Principal Investigator)
Milwaukee, Wisconsin

Koppers Company, Incorporated
Activated Sludge Treatment of Wood Preserving Waste Water
X. P. Laskaris (Principal Investigator)
Carbondale, Illinois

State Department of Environmental Conservation
High Performance Bio-Treatment of Municipal Sewage
C. Beer (Principal Investigator)
Albany, New York

Chino City Government
Reclamation of Wastewater by Controlled Biological Kinetics
D. Feuerstein (Principal Investigator)
Chino, California

U.S. Environmental Protection Agency
National Environmental Research Center
Installation of Instrumentation and Control Devices
for Three-Stage Activated Sludge
W. W. Schuk (Principal Investigator)
Cincinnati, Ohio

Wyoming City Government
Disinfection of Activated Sludge Effluent
J. A. Sheeran (Principal Investigator)
Grand Rapids, Michigan

U.S. District of Columbia Government
Department of Environmental Services
BOD Removal, Nitrification and Denitrification in
a Single Stage Activated Sludge System
A. B. Hais (Principal Investigator)
Washington, D.C.

U.S. Environmental Protection Agency
National Environmental Research Center
A Three-Stage Activated Sludge System for Removing
Virus and Heavy Metals
S. Roan (Principal Investigator)
Cincinnati, Ohio

University of Connecticut
Department of Chemical Engineering
Addition of Powdered Activated Carbon to Activated
Sludge Reactors
D. W. Sundstrum and H. E. Klei (Principal Investigators)
Storrs, Connecticut

University of Connecticut
Department of Agricultural Economics
Efficient Pricing for Urban Waste Water Renovation
R. L. Leonard and R. Laak (Principal Investigators)
Storrs, Connecticut

University of California
Department of Chemical Engineering
Size of Bacterial Aggregations on Rates of Growth
and Bio-Oxidation
A. P. Jackman (Principal Investigator)
Davis, California

Beaunit Fibers
Activated Sludge Treatment of Nylon Wastewaters
Using Enriched Air
R. K. Priest (Principal Investigator)
Etowah, Tennessee

Lehigh University
Center for Surfacing and Coating Research
Effect of Various Additives on the Improved
Dewatering of Activated Sludge
F. J. Micale (Principal Investigator)
Bethlehem, Pennsylvania

Purdue University
Joint Highway Research Project
Treatment of Sanitary Wastes at Interstate Rest Areas
J. E. Etzel and A. J. Steffen (Principal Investigators)
Lafayette, Indiana

U.S. Environmental Protection Agency
National Environmental Research Center
Determination of the Effects of Doses of Sludge on
the Activated Sludge Process
B. V. Salotto (Principal Investigator)
Cincinnati, Ohio

New York City Department of Water Resources
Demonstration of the Oxygen Aeration Process to
Upgrade Existing Waste Treatment Plants
W. Pressman (Principal Investigator)
New York, New York

Los Angeles County Sanitation District 2
Operation of Two-Stage Activated Sludge System
J. Gasser (Principal Investigator)
Los Angeles, California

Los Angeles Board of Public Works
Characterization of the Activated Sludge Process
at the Hyperion Treatment Plant
A. D. Leipzig (Principal Investigator)
Los Angeles, California

U.S. Environmental Protection Agency
National Environmental Research Center
Microscopic Examination of and Characterization of
Batch Flux Settling Properties on Activated Sludges
R. F. Lewis (Principal Investigator)
Cincinnati, Ohio

U.S. Environmental Protection Agency
National Environmental Research Center
Investigate the Effectiveness of Process Control by
Computation
R. Smith (Principal Invetigator)
Cincinnati, Ohio

U.S. Environmental Protection Agency
National Environmental Research Center
Identification of Pollutants in Petroleum Refinery Waste Waters After Activated Sludge Treatment
L. H. Keith (Principal Investigator)
Corvallis, Oregon

Hydrotechnic Corporation
High Rate Deep Bed Filtration of Activated Sludge Plant Effluent
P. J. Harvey (Principal Investigator)
New York, New York

Michigan State University
Institute of Water Research
The Characterization of Soluble Phosphorus Compounds in Biological Waste Treatment Processes
F. M. Ditri (Principal Investigator)
East Lansing, Michigan

University of Puerto Rico
Department of Civil Engineering
Relationship Between BOD Removal and LAS Detergent Removal
N. Tang (Principal Investigator)
Mayaguez, Puerto Rico

University of Kentucky
Department of Chemical Engineering
Process Control of Activated Sludge Treatment
R. I. Kermode and R. W. Brett (Principal Investigators)
Lexington, Kentucky

U.S. Army Plans and Studies Directorate
Emission of Viruses from Sewage Treatment Plant Facilities
C. J. Spendlove and P. A. Adams (Principal Investigators)
Dugway Proving Ground
Dugway, Utah

University of Texas
Department of Civil Engineering
Hydrocarbons Emitted During Aeration of Refinery Wastewater
J. O. Ledbetter and J. W. Kim (Principal Investigators)
Austin, Texas

University of Maine
Department of Civil Engineering
Adsorption as a Protective Mechanism for Water-Borne Viruses
O. J. Sproul (Principal Investigator)
Orono, Maine

SECONDARY TREATMENT: TRICKLING FILTERS

Research programs currently being conducted are being performed by the following organizations:

Georgia Institute of Technology
Environmental Resources Center
Bark as a Trickling-Filter Dewatering Media for Pulp and Paper Mill Sludge
G. R. Lightsey (Principal Investigator)
Atlanta, Georgia

Georgia Institute of Technology
Department of Civil Engineering
Sanitary Landfill Leachate Treatment by Trickling Filter
F. G. Pohland (Principal Investigator
Atlanta, Georgia

Harris Company Water Control and Improvement
Removal of BOD_5, Suspended Solids, Nitrogen, and Phosphorous by Trickling Filter
N. O. Swennes
Seabrook, Texas

U.S. Army Plans and Studies Directorate
Emission of Viruses from Sewage Treatment Plant Facilities
C. J. Spendlove and P. A. Adams (Principal Investigators)
Dugway, Utah

SECONDARY TREATMENT: AERATED LAGOONS

There is a significant amount of research presently underway which is directed towards industrial waste applications of aerated lagoons and upgrading of lagoon effluents through tertiary treatment. Research is being conducted by the following organizations:

University of Arizona
Department of Civil Engineering
Tertiary Treatment of Aerated Lagoon Effluent by Soil Filtration to Permit Reuse for Park and Playground Irrigation
Tucson, Arizona

U.S. Environmental Protection Agency
National Environmental Research Center
Organic Pollutants in Refinery Wastewater Following Aerated Lagoon Treatment
L. H. Keith (Principal Investigator)
Corvallis, Oregon

University of Texas
Department of Civil Engineering
Hydrocarbons Emitted During Aeration of Refinery Wastewater
J. O. Ledbetter and J. W. Kim (Principal Investigators)
Austin, Texas

University of Washington
Department of Forest Products
Natural, Acidic, and Water Soluble Compounds Resulting from Aerated Lagoon Treatment
B. F. Hruthfiord (Principal Investigator)
Seattle, Washington

Auburn University
Department of Agricultural Engineering
Pollutional Characteristics of Swine Waste following Treatment in a Two-Stage, Anaerobic-Aerobic Lagoon System
R. E. Hermanson and J. L. Koon (Principal Investigators)
Auburn, Alabama

Utah State University
Utah Center for Water Resources Research
Performance Evaluation of an Existing Lagoon
J. H. Reynolds and E. J. Middlebrooks (Principal Investigators)
Logan, Utah

Clemson University
Department of Horticulture
Treatment of Wastewaters from Food Processing Plants by Aerated Lagoon
L. Vanbularicom and T. L. Senn (Principal Investigators)
Clemson, South Carolina

Crown Zellerbach Corporation
Coliform Organisms in a Full-Scale Aerated Stabilization Basin
H. Amberg (Principal Investigator)
Camas, Washington

Kent Cheese Company
The Effectiveness of Aerated Lagoons for the Treatment of Industrial Wastewater
F. R. Paul (Principal Investigator)
Melrose Park, Illinois

University of Texas
Department of Civil Engineering
Hydrocarbons Emitted During Aeration of Refinery Wastewater
J. O. Ledbetter and J. W. Kim (Principal Investigators)
Austin, Texas

University of Washington
Department of Forest Products
Natural, Acidic, and Water Soluble Compounds Resulting from Aerated Lagoon Treatment
B. F. Hruthfiord (Principal Investigator)
Seattle, Washington

Auburn University
Department of Agricultural Engineering
Pollutional Characteristics of Swine Waste following Treatment in a Two-Stage, Anaerobic-Aerobic Lagoon System
R. E. Hermanson and J. L. Koon (Principal Investigators)
Auburn, Alabama

Utah State University
Utah Center for Water Resources Research
Performance Evaluation of an Existing Lagoon
J. H. Reynolds and E. J. Middlebrooks (Principal Investigators)
Logan, Utah

Clemson University
Department of Horticulture
Treatment of Wastewaters from Food Processing Plants by Aerated Lagoon
L. Vanbularicom and T. L. Senn (Principal Investigators)
Clemson, South Carolina

Crown Zellerbach Corporation
Coliform Organisms in a Full-Scale Aerated Stabilization Basin
H. Amberg (Principal Investigator)
Camas, Washington

Kent Cheese Company
The Effectiveness of Aerated Lagoons for the Treatment of Industrial Wastewater
F. R. Paul (Principal Investigator)
Melrose Park, Illinois

W. E. Reeves Packinghouse
The Suitability of Anaerobic-Aerobic Lagoons in Meatpacker Waste Treatment Systems
W. E. Reeves (Principal Investigator)
Ada, Oklahoma

East Central State College
Department of Environmental Science
Aerated Aerobic Oxidation Pond Treatment of Packinghouse Wastes
R. H. Ramsey (Principal Investigator)
Ada, Oklahoma

SECONDARY TREATMENT: ANAEROBIC LAGOONS

Research concerning anaerobic lagoons is currently being conducted by the following organizations:

W. E. Reeves Packinghouse
Demonstration of the Suitability of an Anaerobic-Aerobic Lagoon System for Treating Meat Packer Wastes
W. E. Reeves (Principal Investigator)
Ada, Oklahoma

University of Georgia
Department of Agricultural Engineering
Design and Operation Criteria for an Anaerobic Lagoon for Swine Waste Treatment
R. E. Smith (Principal Investigator)
Athens, Georgia

Auburn University
Department of Agricultural Engineering
A Swine Waste Treatment System Consisting of a Series Connected Anaerobic Lagoon, Aerated Lagoon or Oxidation Ditch, and Final Aerobic Lagoon
R. E. Hermanson and K. L. Koon (Principal Investigators)

Utah State University
Utah Center for Water Resources Research
Performance Evaluation of an Existing Lagoon
J. H. Reynolds and E. J. Middlebrooks (Principal Investigators)
Logan, Utah

SECONDARY TREATMENT: PONDING

Several current research projects concerning the removal of contaminants by lagooning procedures are listed below:

University of Kentucky
Department of Civil Engineering
Tertiary Treatment Using Oxidation Ponds
R. A. Lauderdale (Principal Investigator)
Lexington, Kentucky

Asian Institute of Technology
Nitrogen Uptake through the Regrowth of Algae in Secondary Ponds
M. G. McGarry and S. Pinkayan (Principal Investigators)
Bangkok, Thailand

University of Texas
Department of Civil Engineering
The Presence of Pathogenic Organisms Including Virus and the Susceptibility of the Effluent in Stabilization Ponds to Chemical Disinfection
J. F. Malina (Principal Investigator)
Austin, Texas

University of North Carolina
Department of Environmental Sciences and Engineering
Suspended Solids Removal in Settling Ponds
J. C. Brown (Principal Investigator)
Chapel Hill, North Carolina

Gorgas Memorial Institute
Stabilization Pond Operation in Tropical Areas
M. D. Young and K. W. Langley (Principal Investigators)
Washington, D. C.

U.S. Environmental Protection Agency
National Environmental Research Center
Organic Pollutant Discharge in Industrial Wastewater Following Lagoon Treatment
L. H. Keith (Principal Investigator)
Corvallis, Oregon

TERTIARY TREATMENT: FILTRATION

The removals of various contaminants have been the subject of several research projects because of the increasing interest in filtration as a process for tertial wastewater treatment. Organizations performing this research include the following:

Dallas City Water Utilities Department
Heavy Metals Removal Using a Combination of Processes Including Dual Media Filtration
H. W. Wolf (Principal Investigator)
Dallas, Texas

Colorado State University
Agricultural Experiment Station
Reduction of COD and BOD from Animal Feed Lot Runoff by Ultrafiltration
S. M. Morrison and R. P. Martin (Principal Investigators)
Fort Collins, Colorado

University of Illinois
Department of Civil Engineering
The Role of Polymers in the Removal of Viruses by Filtration
R. S. Englebrecht and D. Amirhor (Principal Investigators)
Urbana, Illinois

Water Pollution Research Laboratory
Department of the Environment
Fundamentals of Biological Filtration Mechanisms
Stevenage, England, United Kingdom

University of Wyoming
Department of Civil and Architectural Engineering
The Effectiveness of Sand Filters for Removal of Colloidal Manganese Oxides when Selected Cations are Used as Chemical Conditioners
S. R. Jenkins (Principal Investigator)
Laramie, Wyoming

U.S. Environmental Protection Agency
National Environmental Research Center
Identification of Organic and Inorganic Compounds Remaining in Physical-Chemical Treated Municipal Wastes
A. W. Garrison (Principal Investigator)
Corvallis, Oregon

U.S. Department of Agriculture
Western Regional Research Center
Evaluation of the Uni-Flow Filter System for the Separation of Solids from Food Processing Waters and Waste Effluents
K. Popper (Principal Investigator)
Berkeley, California

Asian Institute of Technology
Filtration Processes in Tertiary Treatment Under Tropical Conditions
M. G. McGarry and S. Pinkayan (Principal Investigators)
Bangkok, Thailand

Minneapolis-St. Paul Sewer Board
Physical-Chemical Treatment Evaluation Including Multi-Media Filtration
R. M. Susag (Principal Investigator)
St. Paul, Minnesota

Woods Hole Oceanographic Institute
Department of Biology
Virus and Bacteria Removal from Secondary Effluent by Photoinactivation and Sand Filtering
B. H. Ketchum (Principal Investigator)
Woods Hole, Massachusetts

Hatfield Township Municipal Authority
BOD_5, COD, Suspended Solids and Phosphorus Removal by Filtration
T. Greenland (Principal Investigator)
Colmar, Pennsylvania

U.S. Environmental Protection Agency
National Environmental Research Center
Removal of Toxic Metals by Filtration
S. A. Hannah (Principal Investigator)
Cincinnati, Ohio

University of Delaware
Department of Biological Sciences
The Use of Blue-Green Algal Viruses as Indicators of the Efficiency of Water Treatment in Elimination of Human Enteric Viruses
D. S. Herson and J. Nobleharvey (Principal Investigators)
Newark, Delaware

University of North Carolina
Department of Environmental Sciences and Engineering
Tertiary Solids Removal by Filtration
J. C. Brown (Principal Investigator)
Chapel Hill, North Carolina

J. P. Stevens and Company, Incorporated
BOD_5 and Suspended Solids Removal by Filtration
W. R. Hogue (Principal Investigator)
Greensboro, North Carolina

Roy F. Weston, Incorporated
Treatment of Storm Runoff by Filtration
R. Weston (Principal Investigator)
Westchester, Pennsylvania

Fram Corporation
Feasibility Investigation of Self-Cleaning Strainer and Filter
S. S. Blecharczyk (Principal Investigator)
Providence, Rhode Island

U.S. Environmental Protection Agency
National Environemntal Research Center
Chlorination and Filtration of Effluent from Denitrification through Dual or Tri Media Filters
D. F. Bishop (Principal Investigator)
Cincinnati, Ohio

American University
Department of Chemistry
Identification of Trace Organics Following Lime Clarification, Filtration and Carbon Adsorption
M. H. Aldridge and T. A. Pressley (Principal Investigators)
Washington, D.C.

State Department of Iron Range Resources
Peat and Peat Soil Mixtures as Filter Media
R. Scuffy (Principal Investigator)
St. Paul, Minnesota

Auburn University
Department of Civil Engineering
The Effectiveness of Sand Filters for the Removal of Specific Viruses from Water Using Selected Cations as Filter Aids
S. R. Jenkins (Principal Investigator)
Auburn, Alabama

University of Maine
Department of Civil Engineering
Controlled Filtration Experiments Using Uniform Sand Columns and Chemically Conditioned Influent Suspensions of Latex Monodispersions and Polydispersions, Secondary Effluents and Virus Suspensions
M. M. Ghosh and W. F. Brutsaert (Principal Investigators)
Orono, Maine

TERTIARY TREATMENT: ADSORPTION

The removal by carbon adsorption of certain groups of contaminants has been the topic of many research projects. This interest arises from the fact that carbon adsorption offers promise in the tertiary removal of troublesome contaminants (i.e., trace organics) in municipal wastewater. Organizations performing this research include the following:

University of Texas
Department of Civil Engineering
Removal of Mercury by a Combination of Chelation and Activated Carbon Adsorption
M. J. Humenick and R. Austin (Principal Investigators)
Austin, Texas

University of New South Wales
School of Chemical Engineering
Treatment of Refinery Wastes by Carbon Adsorption
K. A. Buckle and C. J. Fell (Principal Investigators)
Sydney, New South Wales, Australia

Dallas City Water Utilities Department
Removal of Heavy Metals Using a Combination of Treatment Processes Including Carbon Adsorption
H. W. Wolf (Principal Investigator)
Dallas, Texas

U.S. Environmental Protection Agency
National Environmental Research Center
The Fate of As, Ba, Cd, Se, Cr, and Hg During Activated Carbon Treatment
J. M. Symons (Principal Investigator)
Cincinnati, Ohio

U.S. Environmental Protection Agency
National Environmental Research Center
Identification of Trace Organic and Inorganic Compounds Remaining in Physical-Chemical Treated Municipal Wastes
A. W. Garrison (Principal Investigator)
Corvallis, Oregon

University of Illinois
Department of Civil Engineering
Removal of Organics from Military Waste Streams Using Ozone and Activated Carbon
E. S. Chain (Principal Investigator)
Urbana, Illinois

Asian Institute of Technology
Activated Carbon Adsorption of Color
M. G. McGarry and S. Pinkayan (Principal Investigators)
Bangkok, Thailand

Minneapolis-St. Paul Sewer Board
Suspended Solids, BOD_5, Ammonia, and Phosphate Removal by Carbon Adsorption
R. H. Susag (Principal Investigator)
St. Paul, Minnesota

U.S. Environmental Protection Agency
National Environmental Research Center
Process Modifications Using Activated Carbon Adsorption
S. A. Hannah (Principal Investigator)
Cincinnati, Ohio

U.S. Environmental Protection Agency
National Environmental Research Center
Removal of Synthetic Organic Compounds from Wastewater by Activated Carbon
R. A. Dobbs (Principal Investigator)
Cincinnati, Ohio

Infilco Fuller Company
Activated Carbon Treatment in Slurry Clarifiers
C. F. Garland (Principal Investigator)
Tucson, Arizona

American University
Department of Chemistry
Characterization of Trace Oranics in Wastewater Following Carbon Adsorption
M. M. Aldridge and T. A. Pressley (Principal Investigators)
Washington, D.C.

University of Connecticut
Department of Chemical Engineering
The Addition of Powdered Activated Carbon to Activated Sludge Reactors
D. E. Sundstrom and H. E. Klei (Principal Investigators)
Storrs, Connecticut

TERTIARY TREATMENT: CHEMICAL TREATMENT

A considerable amount of research is currently being performed concerning the removal of the various public health impairing contaminants by chemical treatment processes. The organizations conducting this research include the following:

Life Systems, Incorporated
Use of the Electrochemical Process to Disinfect Water and Wastewater in Reuse Systems
R. A. Wynveen (Principal Investigator)
Cleveland, Ohio

University of Illinois
Department of Civil Engineering
Coagulation of Wastewaters Using Fly Ash
J. T. O'Connor and J. H. Gulledge (Principal Investigators)
Urbana, Illinois

Water Pollution Research Laboratory
Physico-Chemical Processes to Remove Ammonia, Phosphate, and Organic Matter from Sewage Effluent
Department of the Environment
Stevenage, England, United Kingdom

U.S. Environmental Protection Agency
National Environmental Research Center
Study of Treatment Processes for the Removal of Trace Metals and Nitrates
J. M. Symons (Principal Investigator)
Cincinnati, Ohio

U.S. Department of the Interior
Metallurgy Research Center
Treatment of Metallurgical Wastes by Chemical Coagulation
A. A. Cochran and L. C. George (Principal Investigators)
Rolla, Missouri

U.S. Environmental Protection Agency
National Environmental Rsearch Center
Identification of Pollutants in Physical-Chemical Treated Wastes
A. W. Garrison (Principal Investigator)
Corvallis, Oregon

Northeastern University
Department of Civil Engineering
Effects of Interactions of Chemicals for Water Treatment
J. J. Cochrane and W. Glover (Principal Investigators)
Boston, Massachusetts

Rex Chainbelt, Incorporated
Technical Center
Chemical Oxidation in the Treatment of Combined Sewer Overflows
W. J. Katz (Principal Investigator)
Milwaukee, Wisconsin

U.S. National Aeronautics and Space Administration
Urban Systems Project Office
Chemical Coagulation for the Removal of Suspended Solids
J. Poradek (Principal Investigator)
Houston, Texas

Minneapolis-St. Paul Sewer Board
Physical-Chemical Treatment Plant Evaluation
R. H. Susag (Principal Investigator)
St. Paul, Minnesota

Fairfax County Department of Public Works
Reduction of BOD_5, Nitrogen, Phosphorus and Suspended Solids by Chemical Treatment
J. E. Sunday (Principal Investigator)
Fairfax, Virginia

U.S. Environmental Protection Agency
National Environmental Research Center
Physical-Chemical Processes to Remove Potentially Hazardous Compounds from Wastewater
R. A. Dobbs (Principal Investigator)
Cincinnati, Ohio

U.S. Environmental Protection Agency
National Environmental Research Center
Removal of Toxic Metals by Physical-Chemical Treatment
S. A. Hannah (Principal Investigator)
Cincinnati, Ohio

University of Delaware
Department of Biological Sciences
The Use of Blue-Green Algal Viruses as Indicators of the Efficiency of Coagulation in Elimination of Human Enteric Viruses
D. S. Herson and J. Nobleharvey (Principal Investigators)
Newark, Delaware

Roy F. Weston, Incorporated
Treatment of Urban Storm Runoff by Coagulation
R. Weston (Prinicipal Investigator)
West Chester, Pennsylvania

Grumman Aerospace Corporation
Chemical Coagulation in the Treatment and Recovery of Fluoride Industrial Wastes
C. Staebler (Principal Investigator)
Bethpage, New York

U.S. Environmental Protection Agency
National Environmental Research Center
Analysis of Organic Pollutants in Industrial Wastewater Following Chemical/Biological Treatment
L. H. Keith (Principal Investigator)
Corvallis, Oregon

Dallas City Water Utilities Department
Removal of Heavy Metals by Chemical Treatment Processes
H. W. Wolf (Principal Investigator)
Dallas, Texas

American University
Department of Chemistry
Identification of Trace Organics in Physical-Chemical Treatment Effluents
M. H. Aldridge and T. A. Pressley (Principal Investigators)
Washington, D.C.

University of Maine
Department of Chemical Engineering
Chemical and/or Chemico-Mechanical Modification of Sludges
E. G. Bobalek and R. E. Durst (Principal Investigators)
Orono, Maine

Iowa State University
Department of Civil Engineering
Chemical Treatment for Phosphorus Removals
E. R. Baumann and C. S. Oulman (Principal Investigators)
Ames, Iowa

New York State Agricultural Experiment Station
Physical-Chemical Treatment of Refractory Food Processing Liquid Wastes
R. H. Walter (Principal Investigator)
Geneva, New York

R.A.I. Research Corporation
Technical and Economic Feasibility Study of Electrochemical Oxidation of Whey
S. B. Tuwiner (Principal Investigator)
Long Island City, New York

Aerojet General Corporation
Department of Treatment Systems
Development of Treatment Process for Chlorinated Hydrocarbon Pesticide Manufacturing and Processing Wastes
K. H. Sweeny (Principal Investigator)
El Monte, California

University of Rhode Island
Department of Civil and Environmental Engineering
The Electrochemical Process for Nutrient Removal
C. P. Poon (Principal Investigator)
Kingston, Rhode Island

San Antonio Department of Public Works
Demonstration of Lime Coagulation for Removal of Virus from Municipal Sewage
H. C. Norris (Principal Investigator)
San Antonio, Texas

U.S. District of Columbia Government
Department of Environmental Services
Stack Gas Analysis Studies for Air Pollution Control from Incineration of Wastewater Sludge
A. B. Hais (Principal Investigator)
Washington, D.C.

TERTIARY TREATMENT: ION EXCHANGE

Several research programs are currently being conducted in regards to the application of ion exchange in municipal wastewater treatment. This research is being performed by the following organizations:

U.S. Department of the Interior
Metallurgy Research Center
Treatment of Metallurgical Wastes Using Ion Exchange
A. A. Cochran and L. C. George (Principal Investigators)
Rolla, Missouri

U.S. Department of Agriculture
Western Regional Research Center
Separation of Liquids from Solids in Food Processing Wastes Using Ion Exchange
K. Popper (Principal Investigator)
Berkeley, California

Grumman Aerospace Corporation
Treatment and Recovery of Fluoride Industrial Wastes Using Ion Exchange
C. Staebler (Principal Investigator)
Bethpage, New York

Louis Koenig Research
Operations Research for Advanced Waste Treatment Process Including Ion Exchange
L. Koenig (Principal Investigator)
San Antonio, Texas

Minneapolis-St. Paul Sewer Board
Suspended Solids, BOD_5, Ammonia, and Phosphate Removal by Ion Exchange
R. H. Susag (Principal Investigator)
St. Paul, Minnesota

TERTIARY TREATMENT: NITRIFICATION-DENITRIFICATION

The following organizations are currently conducting research programs on this subject:

Hatfield Township Municipal Authority
Analysis of an Integrated Series of Unit Processes Including Biological Nitrification
T. Greenlund (Principal Investigator)
Colmar, Pennsylvania

Harris Company Water Control and Improvement
Suspended Growth Nitrification and Column Denitrification in Trickling Filters
N. O. Swennes (Principal Investigator)
Seabrook, Texas

U.S. Environmental Protection Agency
National Environmental Research Center
Nitrification and Denitrification by a Single Stage Activated Sludge Process
D. F. Bishop (Principal Investigator)
Cincinnati, Ohio

U.S. Environmental Protection AGency
National Environmental Research Center
Staged Nitrification-Denitrification System for Removing Virus and Heavy Metals
S. Roan (Principal Investigator)
Cincinnati, Ohio

U.S. District of Columbia Government
Department of Environmental Services
Nitrification and Denitrification by a Single Stage Activated Sludge System
A. B. Hais (Principal Investigator)
Washington, D.C.

DISINFECTION: CHLORINATION

Research is being performed by the following organizations:

Holifield National Laboratory
Environmental Sciences Division
Chlorinated Organic Compounds Present in Natural Waters
C. W. Gehrs and W. W. Pitt (Principal Investigators)
Oak Ridge, Tennessee

Dallas City Water Utilities Department
Removal of Heavy Metals by Chlorination
H. W. Wolf (Principal Investigator)
Dallas, Texas

U.S. Environmental Protection Agency
National Environmental Research Center
Identification of Chlorinated Organics in Chlorinated Waste Activated Sludge
A. W. Garrison (Principal Investigator)
Corvallis, Oregon

Northeastern University
Department of Civil Engineering
Effects of Interactions of Chemicals for Water Treatment
J. J. Cochrane and W. Glover (Principal Investigators)
Boston, Massachusetts

University of Tennessee
Farm Electrification Research Branch
Tests of Chlorinators on Water from Typical Farm Sources and on Septic Tank Effluent
R. B. Stone (Principal Investigator)
Knoxville, Tennessee

University of Hawaii
Department of Public Health Science
Characterization of Refractory Organics of Possible Carcinogenic Significance in Recycled Wastewater
N. C. Burbank and R. E. Green (Principal Investigators)
Honolulu, Hawaii

Asian Institute of Technology
Chlorination Processes as Applied to Tertiary Treatment for the Reclamation of Drinking Water from Sewage
M. G. McGarry and S. Pinkayan (Principal Investigators)
Bangkok, Thailand

Koppers Company, Incorporated
Pre- and Post-Chlorination
X. P. Laskaris (Prinicipal Investigator)
Carbondale, Illinois

Philadelphia Water Department
Chlorination Treatment of Combined Sewer Overflow
C. F. Guarino (Principal Investigator)
Philadelphia, Pennsylvania

Minneapolis-St. Paul Sewer Board
Evaluation of Chlorination Treatment for Disinfection
R. H. Susag (Principal Investigator)
St. Paul, Minnesota

U.S. Environmental Protection Agency
National Environmental Research Center
Bactericidal Effect of Various Combinations of Gamma Radiation and Chloramine on Aquaeous Suspension of *Escherichia coli*
A. D. Venosa (Principal Investigator)
Cincinnati, Ohio

Fairfax County Department of Public Works
Transportable Advanced Wastewater Treatment Plant for Interim Use
J. E. Sunday (Principal Investigator)
Fairfax, Virginia

Wyoming City Government
Ozonation and Chlorination with Dechlorination of Chlorinated Effluent
J. A. Sheeran (Principal Investigator)
Grand Rapids, Michigan

University of Delaware
Department of Biological Sciences
The Use of Blue-Green Algae Viruses as Indicators of the Efficiency of Water Treatment on Elimination of Human Enteric Viruses
D. S. Herson and J. Nobleharvey (Principal Investigators)
Newark, Delaware

U.S. Environmental Protection Agency
National Environmental Research Center
Chlorination and Filtration of Effluent from Denitrification
D. F. Bishop (Principal Investigator)
Cincinnati, Ohio

U.S. Environmental Protection Agency
National Environmental Research Center
Use of the Membrane Filter for Chlorinated Effluents
R. A. Greene (Principal Investigator)
Cincinnati, Ohio

Dallas City Water Utilities Department
Removal of Heavy Metals by Chlorination
H. W. Wolf (Principal Investigator)
Dallas, Texas

Dow Chemical Company
Environmental Research Laboratory
Ultraviolet Chlorination of Organic Acids in Waste Brines
R. F. Wukasch (Principal Investigator)

University of Cincinnati
Department of Civil and Environmental Engineering
Inactivation of Viruses in Wastewater by Chlorine and Chlorine Compounds
P. V. Scarpino (Principal Investigator)
Cincinnati, Ohio

University of Minnesota
Department of Chemistry
Chlorination and Ozonation Products of Municipal Sewage and their Environmental Impact
R. M. Carlson (Principal Investigator)
Minneapolis, Minnesota

University of Hawaii
Department of Environmental Health
Use of Bromine Chloride as a Viricide in Hawaii
N. C. Burbank and P. C. Loh (Principal Investigators)
Honolulu, Hawaii

University of Illinois
Department of Civil Engineering
New Microbial Indicator of Wastewater Chlorination Efficiency
R. S. Engelbrecht (Principal Investigator)
Urbana, Illinois

University of Connecticut
Department of Biology
Effects of Chlorine Treatment on Organic Molecules
R. P. Collins (Principal Investigator)
Storrs, Connecticut

University of Maryland
Department of Agricultural Engineering
Chlorination of Runoff from Livestock Operations as a Means to Improve the Bacteriological Quality of the Effluent
J. A. Merkel (Principal Investigator)
College Park, Maryland

U.S. Environmental Protection Agency
National Environmental Research Center
The Fate of a Simulated Marine Environment of Chlorinated Compounds Produced by Wastewater Chlorination
M. A. Horton (Principal Investigator)
Corvallis, Oregon

DISINFECTION: OZONATION

Recent evidence suggesting that chlorinated effluents adversely affect aquatic life has increased the research interest in the possibilities of ozonation. Research is presently being conducted by the following organizations:

University of Louisville
Department of Biology
Enteric Virus Survival in Package Plants and the Upgrading of the Small Treatment Plants Using Ozone
L. S. Cronholm and J. L. Pavoni (Principal Investigators)
Louisville, Kentucky

University of Tennessee
Farm Electrification Research Branch
Ozone Generators for Purification of Farm Water Supplies
R. B. Stone (Prinicipal Investigator)
Knoxville, Tennessee

University of Hawaii
Department of Public Health Science
Characterization of Refractory Organics of Possible Carcinogenic Significance in Recycled Wastewaters
N. C. Burbank and R. E. Green (Principal Investigators)
Honolulu, Hawaii

Philadelphia Water Department
Ozonation Applied to the Treatment of Combined Sewer Overflow
C. F. Guarino (Principal Investigator)
Philadelphia, Pennsylvania

Wyoming City Government
Ozonation and Chlorination with Dechlorination of Chlorinated Effluent
J. A. Sheeran (Principal Investigator)
Grand Rapids, Michigan

University of Illinois
Department of Civil Engineering
Virus Inactivation by Ozone
J. T. O'Connor and T. R. Begley (Principal Investigators)
Urbana, Illinois

Case Western Reserve University
Department of Chemical Engineering
Ozone Applications for the Treatment of Pulp and Paper Mill Effluents
P. B. Melnyk (Principal Investigator)
Cleveland, Ohio

University of Vermont
Department of Biology
The Effectiveness of Ozone in the Oxidation of Lipids (Fats) in Sewage Effluent Water
T. Sproston and J. J. Stoveken (Principal Investigators)
Burlington, Vermont

Montana State University
Department of Chemistry
An Evaluation of Tertiary Water Treatment Via Ozonation
W. L. Waters and A. Rollin (Principal Investigators)
Bozeman, Montana

University of Illinois
Department of Civil Engineering
Removal of Organics Using Ozone
E. S. Chain (Prinicipal Investigator)
Urbana, Illinois

Westgate Research Corporation
U V - Ozone Water Oxidation/Sterilization Process
J. D. Zeff (Principal Investigator)
Venice, California

Aerojet General Corporation
Department of Treatment Systems
Development of Treatment Process for Chlorinated Hydrocarbon Pesticide Manufacturing and Processing Wastes
K. H. Sweeney (Principal Investigator)
El Monte, California

DISINFECTION: RADIATION TREATMENT

Research into the effect of radiation on any of the public health impairing contaminants is being conducted by the following organizations:

University of Hawaii
Department of Public Health Science
Characterization of Refractory Organics of Possible Carcinogenic Significance in Recycled Wastewater
N. C. Burbank and R. E. Green (Principal Investigators)
Honolulu, Hawaii

U.S. Environmental Protection Agency
National Environmental Research Center
Bactericidal Effect of Various Combinations of Gamma Radiation and Chloramine on Aqueous Suspension of *Escherichia coli*
A. D. Venosa (Principal Investigator)
Cincinnati, Ohio

U.S. Army
Engineering Research and Development Laboratories
Radiation Treatment of Wastewater
A. J. Vandenberg (Principal Investigator)
Fort Belvoir, Virginia

University of Tennessee
Farm Electrification Research Branch
Ultraviolet Irradiation for Purification of Farm Water Supplies
R. B. Stone (Principal Investigator)
Knoxville, Tennessee

Florida Institute of Technology
Department of Science Education
Radiation Treatment of Waste Water for Destruction of Viruses and Bacteria
D. D. Woodbridge and W. R. Garrett (Principal Investigators)
Melbourne, Florida

Massachusetts Institute of Technology
Department of Electrical Engineering
Wastewater Treatment With High Energy Electrons
J. G. Trump and K. A. Wright (Principal Investigators)
Cambridge, Massachusetts

THICKENING AND DEWATERING: GRAVITY SLUDGE THICKENER

Ongoing research in the field of gravity thickening of sewage sludge is being conducted by the following organization:

U.S. District of Columbia Government
Department of Environmental Services
Thickening of Wastewater Sludge
A. B. Hais (Principal Investigator)
Washington, D.C.

THICKENING AND DEWATERING: VACUUM FILTERS

Research is currently being conducted in the field of vacuum filtration of sewage sludge by:

U.S. District of Columbia Government
Department of Environmental Services
Vacuum Filtration of Wastewater Sludges
A. B. Hais (Principal Investigator)
Washington, D.C.

SLUDGE STABILIZATION: ANAEROBIC DIGESTION

Research projects presently being conducted with regard to anaerobic digestion are the following:

Georgia Agricultural Experiment Station
Digestion by Anaerobic Systems of Effluents from Vegetable Processing Operations
W. A. Bough and A. L. Shewfelt (Principal Investigators)
Griffin, Georgia

Dartmouth College
Department of Engineering Science
Anaerobic Digestion and Membrane Separation for the Treatment of Domestic Sewage
H. E. Grethlein (Principal Investigator)
Hanover, New Hampshire

University of Southern California
Department of Engineering
The Effects of Alkalinity on Anaerobic Digestion of Primary Effluent
K. Chen and A. Sycip (Principal Investigators)
Los Angeles, California

SLUDGE STABILIZATION: AEROBIC DIGESTION

Ongoing research in the field of aerobic digestion is being conducted by the following organizations:

Georgia Institute of Technology
Department of Applied Biology
Production and Dispersal of Pathogenic Bacterial and Viral Aerosols by Aeration of Contaminated Waters
R. King and T. W. Kethley (Principal Investigators)
Atlanta, Georgia

Georgia Agricultural Experiment Station
Digestion by Aerobic Systems of Effluents from Vegetable Processing Operations
W. A. Bough and A. L. Shewfelt (Principal Investigators)
Griffin, Georgia

SLUDGE STABILIZATION: CHLORINATION

Research projects are currently being conducted in the field of chlorination of sewage sludge by the following organizations:

University of Hawaii
Department of Public Health Science
Characterization of Refractory Organics of Possible Carcinogenic Significance in Recycled Wastewater
N. C. Burbank and R. E. Green (Principal Investigators)
Honolulu, Hawaii

State Department of Environmental Conservation
High Pressure Chlorination Treatment of Primary Sludge
C. Beer (Principal Investigator)
Albany, New York

University of Massachusetts
Department of Civil Engineering
Sludge Stabilization by High-Dosage Chlorination
T. H. Feng (Principal Investigator)
Amherst, Massachusetts

SLUDGE STABILIZATION: LIME TREATMENT

Research projects are presently being conducted in the field of lime treatment of sewage sludges by the following organization:

FMC Corporation
Process to Remove Carbonaceous, Nitrogenous, and Phosphorous Materials from Anaerobic Digester Supernatant and Related Process Streams
H. A. Oldenkamp (Principal Investigator)
Santa Clara, California

SLUDGE STABILIZATION: HEAT TREATMENT

Research is currently being conducted with regard to heat treatment by the following organizations:

U.S. Environmental Protection Agency
National Environmental Research Center
Treatment of Supernatants from Heat Treatment of Sludge
B. V. Salotto (Principal Investigator)
Cincinnati, Ohio

FMC Corporation
Process to Remove Carbonaceous, Nitrogenous, and Phosphorous Materials from Anaerobic Digester Supernatant and Related Process Streams
H. A. Oldenkamp (Principal Investigator)
Santa Clara, California

University of Texas
Department of Civil Engineering
Inactivation of Enteric Viruses and Bacteria During the Heat Treatment of Sludges
J. F. Malina and R. Cain (Principal Investigators)
Austin, Texas

SLUDGE STABILIZATION: WET AIR OXIDATION

Research projects are currently being conducted in the field of wet air oxidation by the following organizations:

University of Delaware
Department of Chemical Engineering
Wet and Catalytic Oxidation of Suspended and Dissolved Wastes
B. E. Anshus and J. R. Katzer (Principal Investigators)
Newark, Delaware

Rose Hulman Institute of Technology
Department of Environmental Engineering
The Influence of Wet-Air Oxidation on the Nitrogen, Phosphorus, Heavy Metal Content and Attachment in Domestic Sewage Sludge
E. H. Curtis and L. E. Sommers (Principal Investigators)
Terre Haute, Indiana

FINAL DISPOSAL: INCINERATION

Ongoing research in the field of incineration is being conducted by the following organizations:

University of Delaware
Department of Civil Engineering
Gravity Thickening of Sludges as a Batch Process
R. I. Dick and C. Tien (Principal Investigators)
Newark, Delaware

U.S. District of Columbia Government
Department of Environmental Services
Stack Gas Analysis Studies for Air Pollution Control
from Incineration of Wastewater Sludges
A. B. Hais (Principal Investigator)
Washington, D.C.

FINAL DISPOSAL: SANITARY LANDFILL

The migration of public health imparing contaminants as a result of sewage sludge burial in landfills is only beginning to be researched and documented. The following organizations are presently performing contract research work:

SCS Engineers, Inc.
Monitoring and Environmental Impact Evaluation of
Sewage Sludge Disposal in Sanitary Landfills
R. J. Lofy and R. P. Stearns (Principal Investigators)
Long Beach, California

Georgia Institute of Technology
Department of Civil Engineering
Sanitary Landfill Stabilization with Leachate Recycle
F. G. Pohland (Principal Investigator)
Atlanta, Georgia

FINAL DISPOSAL: LAND RECLAMATION

Present research projects identified that are relevant to land reclamation are being conducted by the following organizations:

Metropolitan Sanitary District of Greater Chicago
Agricultural Benefits and Environmental Changes
Resulting from the Use of Digested Sludge on Field
Crops
D. T. Lordi (Principal Investigator)
Chicago, Illinois

University of Texas
Virus Survival in Soils Injected with Municipal Wastewater Treatment Residuals
B. P. Sagik (Principal Investigator)
San Antonio, Texas

Florida Institute of Technology
Radiation Treatment of Wastewater for Land Use
D. D. Woodbridge, W. R. Garrett (Principal Investigators)
Melbourne, Florida

Pennsylvania State University
Effects of Spray Irrigation of Municipal Wastewater on the Rate and Total Accumulation of Heavy Metals
W. E. Sopper (Principal Investigator)
University Park, Pennsylvania

Purdue University
Utilization and Disposal of Municipal, Industrial, and Agricultural Processing Wastes on Land
L. E. Sommer (Principal Investigator)
Lafayette, Indiana

University of Wisconsin
Utilization and Disposal of Municipal, Industrial, and Agricultural Processing Wastes on Land
D. Keeney (Principal Investigator)
Madison, Wisconsin

U. S. Army
Engineer Power Group
Radiation Treatment of Wastewater for Land Use
A. J. Vandenberg (Principal Investigator)
Fort Belvoir, Virginia

ATMOSPHERIC PATHWAYS

Research presently underway on the generation, migration, and/or health effects associated with aerosols from sewage treatment and disposal processes, is reported by the following organizations:

U.S. Army Medical Bioengineering and Development Laboratory
S. A. Schaub (Principal Investigator)
Fort Detrick, Frederick, Maryland

U.S. Army Plans and Studies Directorate
Emission of Viruses from Sewage Treatment Plant Facilities
C. J. Spendlove and P. A. Adams (Principal Investigators)
Dugway Proving Ground
Dugway, Utah

Southwest Research Institute
Department of Chemistry and Chemical Engineering
Health Implications of Sewage Treatment Plant Facilities
D. E. Johnson and J. T. Goodwin (Principal Investigators)
San Antonio, Texas

Georgia Institute of Technology
Department of Applied Biology
Production and Dispersal of Pathogenic Bacterial and Viral Aerosols by Aeration of Contaminated Waters
R. King and T. W. Kethley (Principal Investigators)
Atlanta, Georgia

University of Texas
Department of Civil Engineering
Hydrocarbons Emitted During Aeration of Refinery Wastewater
J. O. Ledbetter and J. W. Kim (Principal Investigators)
Austin, Texas

WASTEWATER TO LAND: GROUNDWATER

Research programs currently underway in this field include:

Pennsylvania State University
McIntire Stennis Program
Effects of Spray Irrigation of Municipal Wastewater on the Rate and Total Accumulation of Heavy Metals
W. E. Sopper (Principal Investigator)
University Park, Pennsylvania

San Antonio Department of Public Works
Demonstration of Virus Removal from Municipal Sewage
H. C. Norris (Principal Investigator)
San Antonio, Texas

Minnesota State Department of Iron Range Resources
and Rehabilitation
Treatment of Wastes Using Peat, and Peat in Combination with Soil
R. Scuffy (Principal Investigator)
St. Paul, Minnesota

University of Texas
Department of Civil Engineering
Soil Treatment of Concentrated Organic Wastewaters
H. Applegate (Principal Investigator)
El Paso, Texas

West Virginia University
Water Research Institute
Effects of Plant Nutrients and Heavy Metals from
Land Application of Sludge
D. J. Horvath and R. N. Singh (Principal Investigators)
Morgantown, West Virginia

University of Wisconsin
Department of Soil Science
Utilization and Disposal of Municipal, Industrial,
and Agricultural Processing Wastes on Land
D. Keeney (Principal Investigator)
Madison, Wisconsin

University of California
Department of Soil Science and Agricultural Engineering
Heavy Metal and Radionuclide Pollution in Relationship
to Crop Plants
A. Wallace (Principal Investigator)
Riverside, California

U.S. Environmental Protection Agency
National Environmental Research Center
Bioactivity of Soils in Land Application of Wastewater
W. R. Duffer (Principal Investigator)
Corvallis, Oregon

U.S. Environmental Protection Agency
Robert S. Kerr Environmental Research Laboratory
Rating Biodegradability of Wastewater Organics in Soil
R. E. Thomas (Principal Investigator)
Ada, Oklahoma

Purdue University
Department of Agronomy
Utilization and Disposal of Municipal, Industrial, and
Agricultural Processing Wastes on Land
L. E. Sommer (Principal Investigator)
Lafayette, Indiana

U.S. Environmental Protection Agency
National Environmental Research Center
Chemical Changes in Wastewater as it Passes Through
Soil Systems
C. G. Enfield (Principal Investigator)
Corvallis, Oregon

U. S. Army
Waterways Experiment Station
The Effects of Land Disposal on Less Permeable Soils
J. Harrison (Principal Investigator)
Vicksburg, Mississippi

University of Texas
Graduate School
Virus Survival in Soils Injected with Municipal Wastewater Treatment Residuals
B. P. Sagik (Principal Investigator)
San Antonio, Texas

California Institute of Technology
Metal Occurrences within Natural Ecosystems
C. C. Patterson (Principal Investigator)
Pasadena, California

East Central State College
Department of Environmental Science
Analysis of a Spray-Runoff Irrigation System
R. H. Ramsey (Principal Investigator)
Ada, Oklahoma

Woods Hole Oceanographic Institute
Department of Biology
Determination of the Most Efficient Vegetation for Nutrient Removal and Water Recharge
B. H. Ketchum (Principal Investigator)
Woods Hole, Massachusetts

Cornell University
Department of Agricultural Engineering
Land Disposal Systems for Animal Wastes
R. C. Loehr (Principal Investigator)
Ithaca, New York

University of Arizona
Department of Civil Engineering
Sand Filtration of Oxidation Pond Effluent for Reuse in Park Irrigation
D. R. Kasper and R. A. Phillips (Principal Investigators)
Tucson, Arizona

State Department of Water Management
The Flora Filter System in the Disposal of Wastewater on Soils
D. Papier (Principal Investigator)
Columbus, Ohio

South Dakota State University
Department of Civil Engineering
Infiltration Lagoons for Tertiary Treatment of Stabilization Pond Effluent
J. N. Dornbush (Principal Investigator)
Brookings, South Dakota

WASTEWATER DISPOSAL TO LAND: CROPS

Research on this topic is currently under investigation by the following organizations:

East Central State College
Department of Environmental Science
Analysis of a Spray-Runoff Irrigation System
R. H. Ramsey (Principal Investigator)
Ada, Oklahoma

Woods Hole Oceanographic Institute
Department of Biology
Determination of the Most Efficient Vegetation for Nutrient Removal and Water Recharge
B. H. Ketchum (Principal Investigator)
Woods Hole, Massachusetts

University of California
Department of Soil Science and Agricultural Engineering
Heavy Metal and Radionuclide Pollution in Relationship to Crop Plants
A. Wallace (Principal Investigator)
Riverside, California

West Virginia University
Water Research Institute
Effects of Plant Nutrients and Heavy Metals from Land Application of Sludge
D. J. Horvath and R. N. Singh (Principal Investigators)
Morgantown, West Virginia

Duke University
Marine Laboratory
The Role of Sewage Effluent and Sludge Disposal in the Introduction of Mercury into Marine and Agricultural Ecosystems
R. T. Barber (Principal Investigator)
Beaufort, North Carolina

Pennsylvania State University
McIntire Stennis Program
Effects of Spray Irrigation of Municipal Wastewater on the Rate and Total Accumulation of Heavy Metals
W. E. Sopper (Principal Investigator)
University Park, Pennsylvania

Northeastern Wisconsin Regional Planning
Use of Emergent Vegetation for the Biological Treatment of Municipal Wastewater
F. L. Spangler and W. R. Duffer (Principal Investigators)
Appleton, Wisconsin

Metropolitan Sanitary District of Greater Chicago
Agricultural Benefits and Environmental Changes Resulting from the Use of Digested Sludge on Field Crops
D. T. Lordi (Principal Investigator)
Chicago, Illinois

WASTEWATER DISPOSAL TO LAND: FRESHWATER SYSTEMS

The following organizations are currently conducting research into the effect on fresh surface water from land disposal of wastewater.

U. S. Army
Waterways Experiment Station
Overland Runoff of Wastewaters
P. G. Hunt (Principal Investigator)
Vicksburg, Mississippi

California Institute of Technology
Department of Geological and Planetary Science
Metal Occurrences within Natural Ecosystems
C. C. Patterson (Principal Investigator)
Pasadena, California

WASTEWATER DISPOSAL TO LAND: MARINE SYSTEMS

Two organizations were discovered which are currently conducting research on the subject topic:

Duke University
Marine Laboratory
The Role of Sewage Effluent and Sludge Disposal in the Introduction of Mercury into Marine and Agricultural Ecosystems
R. T. Barber (Principal Investigator)
Beaufort, North Carolina

California Institute of Technology
Department of Geological and Planetary Science
Metal Occurrences Within Natural Ecosystems
C. C. Patterson (Principal Investigator)
Pasadena, California

SLUDGE DISPOSAL TO LAND: CROPS

On-going research on the agricultural application of sludge is listed below:

Duke University
R. T. Barber (Principal Investigator)
Beaufort, North Carolina

Metropolitan Sanitation District of Chicago
D. T. Lordi (Principal Investigator)
Chicago, Illinois

University of California
A. Wallace (Principal Investigator)
Riverside, California

West Virginia University
D. J. Horvath and R. N. Singh (Principal Investigators)
Morgantown, West Virginia

SCS Engineers
R. J. Lofy and R. P. Stearns (Principal Investigators)
Long Beach, California

SLUDGE DISPOSAL TO WATER: MARINE SYSTEMS

Research projects are presently being conducted by the following organizations with regard to ocean disposal of sludges:

Duke University
Marine Laboratory
The Role of Sewage Effluent and Sludge in the Introduction of Mercury into Marine Ecosystems
R. T. Barber (Principal Investigator)
Beaufort, North Carolina

U.S. Environmental Protection Agency
National Environmental Research Center
Fate of Trace Elements in Placed, Digested Sewage Sludge in an Experimental Site in the New York Bight
M. H. Feldman (Principal Investigator)
Corvallis, Oregon

SLUDGE DISPOSAL TO WATER: SHELLFISH

Research programs currently underway are primarily concerned with the introduction of trace elements from sewage sludges into marine organisms. These programs are listed below:

Duke University
Marine Laboratory
The Role of Sewage Effluent and Sludge in the Introduction of Mercury into Marine Ecosystems
R. T. Barber (Principal Investigator)
Beaufort, North Carolina

U.S. Environmental Protection Agency
National Environmental Research Center
Fate of Trace Metals in Placed, Digested Sewage Sludge in an Experimental Site in the New York Bight
M. H. Feldman (Principal Investigator)
Corvallis, Oregon